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S. J. Niemczyk (Project Coordinator)
K. G. Adams, W. B. Murfin, L. T. Ritchie, E. W. Eppel
J. D. Johnson

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Sandia National Laboratories

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S. J. Niemczyk (Project Coordinator)
K. G Adams
W. B. Murfin
L. T. Ritchie
E. W. Eppel*

J. D. Johnson
The Dikewood Corporation

Date Published: June 1981

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Albuquerque, NM 87185
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Summary

The potential radiological impact of a core-melt accident on the human population has been investigated. In particular, the radiation dose received from radioactivity which could reach the population via liquid pathways has been considered.

Radioactivity could be released directly to the hydrosphere after a core-melt accident as a result of melt-through of the containment basemat followed by any of three processes: (1) leaching of the melt debris; (2) escape of sumpwater through the hole formed by the melt (or from passage out of the contaiment by an alternate route); and (3) depressurization of the containment atmosphere through the melt hole. The three types of releases would differ primarily in their rates, their magnitudes and their radioactive compositions.

Both the containment atmosphere and the sumpwater releases would occur relatively rapidly. However, most of the radionuclides present in these two releases in substantial quantities would be expected to be rather short-lived. Therefore, such releases could have a significant impact at a specific site only if the travel times of the important radionuclides to the human population were small.

In contrast, leaching of radionuclides from the melt debris would be expected to occur relatively slowly. Most of the long-lived isotopes would be expected to be found primarily in the melt debris. Consequently, even though this release occurred relatively slowly, the impact could still be significant.

In contrast to the situation for releases to the atmosphere, accidents corresponding to the most probable RSS (Reactor Safety Study) meltdown categories would result in the largest releases to the hydrosphere. Furthermore, substantial amounts of radioactivity would generally be expected to be released to the hydrosphere during any meltdown accident involving complete melt-through of the containment basemat.

On the basis of subsurface hydrologies alone, sites range from those that essentially preclude any impacts to the human population via liquid pathways to those that can allow significant impacts to occur. Sites with very slow groundwater movement and/or with long distances to nearby surface water tend to be in the former category; sites with relatively rapid groundwater movement and with short distances to nearby surface water or wells usually are in the latter category.

On the basis of surface-water characteristics and the related populations-at-risk, sites range from those having no significant exposure pathways to the human population to sites having several potentially important exposure pathways. Sites with surface waterbodies having rapid initial dilution and fast flushing and/or small potentially exposed human populations would tend to have little net human exposure and, therefore, few adverse impacts; sites with waterbodies having slow dilution and flushing and large potentially-exposed populations would tend to have much more human exposure and, therefore more impacts.

In order to bound the possible radiation doses to humans that could result from release to the hydrosphere following basemat melt-through, calculations were made under assumptions that tend to maximize population doses. Among the conservative assumptions made were the following:

1. Instantaneous release to groundwater of all radioactivity from both sumpwater and melt debris.
2. One-day transit time from the release point to a surface waterbody.
3. No interdiction measures taken to prevent exposure of humans.

The doses reported in the next two paragraphs are believed to be upper bounds on exposures that would occur under realistic conditions.

At approximately 35% of all the reactors considered, the radiation dose to the entire human population as a result of releases to the hydrosphere via the hole in the containment basemat formed by the melt is estimated to be relatively small (in general, much less than 10^4 to 10^5 person-rem). At another 25% of the reactors, the resultant radiation dose is estimated to be no more than 10^6 to 10^7 person-rem, while at the remaining 40%, the potential dose is no larger than 10^7 to 10^8 person-rem for such releases.

The somewhat artificial construct of an "average individual" was used to estimate the individual doses which correspond to the population doses quoted above. It is assumed that this individual is average with respect to usage habits for all the liquid pathways, that all of the individuals usage for the liquid pathways involves the contaminated waterbody, and that the individual lives long enough to be exposed to all the dominant radionuclides. Under these conditions, the to an average individual over an entire lifetime as a result of the releases to the hydrosphere (via the melt hole) is estimated to be less than 1 rem at approximately 60% of all the reactors under discussion. Furthermore, the total dose to such an individual is estimated to be no more than 1 to 10 rem at another 20% of the reactors and no more than 100 rem at the remaining 20% of the reactors.

These percentages for both population and average individual doses are generally thought to be conservative. However, they do not take into account the effect of various design features which might permit the early emergency of radioactivity into the hydrosphere and, hence, result in a significant dose at some site possessing such features. Furthermore, they do not include the effects of direct entry of sumpwater into surface water. In addition, they do not take into consideration the effects of certain variations of both the surface-water hydrologies and the associated populations-at-risk which could result in higher than expected doses.

Variations of conditions at actual sites could easily result in the doses being either higher or lower than those obtained in this (generic) study by at least an order of magnitude. Likewise, reasonable variations in certain portions of the modeling could result in estimated doses which differed from those presented here by an order of magnitude. Thus, the estimated doses and their related percentages-of-occurrence have rather large uncertainties associated with them.

The calculations indicate that if interdictive (mitigating) actions are not taken, then the liquid pathways can perhaps contribute significantly to the risk related to a core meltdown accident. Successful interdiction in the vicinity of the accident site (the radioactive source) could prevent essentially all adverse consequences resulting from the liquid pathways. Interdictive actions taken farther along the pathways could only sometimes effectively prevent all the consequences.

Interdictive procedures taken at the source are always preferable to procedures implemented along the exposure pathways. Relative to pathway procedures source procedures can usually be described as being the following:

easier: the actions required at the source would usually be of a much smaller scale than most pathway procedures;

more effective: mitigating actions taken close to the source could have the potential of completely eliminating the radiation exposure of the human population whereas pathway procedures would never be that effective, except for isolated aquifers;

more permanent: some source procedures would not need to be repeated on as regular basis as some pathway procedures would need to be; and,

less expensive: the actions required could be performed on a very localized scale instead of ove a large part of the affected environment.

Successful source interdiction could be expected to virtually eliminate the radiation dose received by the human population. At a majority of the sites, there would be suffficient time to employ interdictive procedures at the source. However, they would not necessarily always be possible due to both subsurface and engineered features existing at some sites. Furthermore, they would not be possible for direct releases of sumpwater into the accessible portions of the environment.

It is possible that the lack of knowledge of the subsurface regions could hamper any source interdiction procedures performed at some sites. For sites at which relatively rapid action would be necessary, such ignorance could conceivably preclude or at least delay successful source interdiction.

For situations in which source interdictive procedures were not either feasible or successful, then the radiation dose could be mitigated only by actions taken along the exposure pathways. The effectiveness of such procedures would depend upon the characteristics of the affected waterbody(ies).

The Consequences From Liquid Pathways After a Reactor Meltdown Accident

1 Introduction

1.1 Perspective

1.1.1 Purpose

This study investigates one aspect of the potential impact on the human population of a core-melt accident. In particular, it considers the radiation dose from radioactivity which reaches the population via liquid pathways.

Scoping investigations of the liquid pathway consequences to humans were performed for the Reactor Safety Study (RSS).[1] Those investigations indicated that the risks for the hydrospheric pathway were small compared to those for the atmospheric pathway. The study presented here was undertaken to further the level of examination employed in that liquid pathway analysis by extending the methodologies used for the previous study.

Work has been performed to do the following: (1) to quantify the magnitude and the likelihood of releases to the hydrosphere; (2) to establish the ranges and distributions of hydrogeologic properties appropriate to existing and proposed LWR (light water reactor) sites and to describe the corresponding radionuclide flow characteristics; (3) to model transport in the surface water; (4) to estimate the time-dependent transport of radionuclides among the various trophic levels in the aquatic foodweb and to consider the movement of radionuclides along the other liquid exposure pathways; (5) to estimate the population dose and resultant health effects; and (6) to discuss mitigating actions which could be employed to reduce the risk to the public. The risks for the hydrospheric pathway are compared to those given in the RSS for the atmospheric pathway.

1.1.2 The Reactor Safety Study

For the Reactor Safety Study, a generic treatment was performed to assess the risks associated with core-melt accidents. In that study, which was published as WASH-1400, two specific reactor designs were reviewed in detail. Possible accident sequences were delineated for each of them; event trees and fault trees were constructed and estimates of the probabilities of various types of accidents were developed. The accident sequences were grouped into 9 pressurized water reactor (PWR) categories and 5 boiling water reactor (BWR) categories, according to their release magnitudes, times of release, durations of release, warning times for evacuation, heats of release and energy contents of the released atmospheres; in other words, they were grouped according to those features upon which the atmospheric pathway consequences depend. Phenomenological models were used to consider the behavior of the radioactive materials during the different accident sequences and to describe the subsequent transport and movement of radioactive material outside the reactor containment. The magnitudes of the various types of releases and their consequences to the human population were also described for each of the 9 PWR and the 5 BWR accident categories. In addition, the aggregate risks to the population from the various types of potential accidents were estimated.

1.1.3 Other Liquid Pathway Studies

Since the RSS, two studies dealing with just the effects of releases to the liquid pathway have been performed.[2,3] Both of these studies have had as their goal the consideration of the potential environmental impacts of accidental releases of radioactivity to the hydrosphere from floating nuclear plants. In particular, these studies attempted to compare the consequences of releases to the hydrosphere from floating plants to those from land-based nuclear plants. Because of their restricted goals, certain portions of both these studies were carried out in manners which make straightforward comparison to the results of the atmospheric releases considered in the RSS difficult, if not impossible.

The study presented in this report has attempted to circumvent at least some of those problems in the descriptions used in the previous reports which obviate comparison between the previously-estimated consequences for releases to the atmosphere and the consequences for releases to the hydrosphere. Specifically, portions of the atmospheric and hydrospheric pathways which should involve consonant considerations have been treated in such a fashion. For example, the radioactive releases to the hydrosphere have been taken in this study to be the complements of the releases to the atmosphere considered in the RSS.

1.1.4 Relative Importance of Liquid Pathways

Historically, the radiological consequences resulting from accidental releases of material to liquid pathways have not been examined with the same degree of detail as the consequences for releases to the atmospheric pathways. This is because the airborne risk has been taken to be dominant.

For a core-melt accident, the dominance of the airborne risk can be expected for at least two interrelated reasons. First, the initial time scale on which the radioactive species could reach the population would usually be relatively short for the atmospheric pathway. As a result, that radioactivity would tend to be rather concentrated, at least at early times. In contrast, the movement of the radionuclides to accessible portions of the environment via the liquid pathways would typically be relatively slow. Therefore, the radioactivity in the hydrosphere would tend to be comparatively dispersed at the time of contact with the human population. As a result of the relatively high initial concentrations for the atmospheric pathway, at least the early doses to individuals could be much larger than those for the liquid pathways. In addition, because of the small time until initial exposure, many short-lived radionuclides, such as some of the iodine isotopes, could be important for the atmospheric pathway whereas they usually could not be significant for any liquid pathway.

The second reason that the airborne risk can be expected to be dominant is that initial atmospheric exposure would usually be involuntary whereas exposure to at least most of the liquid pathway contamination could often be avoided by choice. Use of radiologically contaminated water and related items might be voluntary if adequate monitoring were employed. As a result, exposure could often be prevented by avoidance of such contamination. Furthermore, actions could frequently be taken to prevent all of the radioactivity released to the hydrosphere from ever reaching important exposure pathways.

Thus, one can say that the airborne risk is expected to be dominant for the human population because the initial atmospheric exposure would not always be avoidable whereas almost all exposure from the liquid pathway could usually be circumvented by appropriate interdictive measures. Consequently, the relative importance of the two pathways is due to the time scales on which the two exposures could initially occur and to the related differences in the electivity of the two types of exposures.

1.1.5 Scope

The scope of this study is essentially the same as that of the consequence modeling in the RSS. However, that study primarily investigated the consequences of atmospheric releases while this study considers the impact of hydrospheric releases (Figure 1.1).

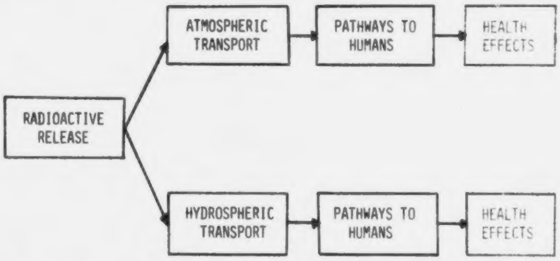

Figure 1.1 Schematic Diagram of the Models Used in the RSS for Calculating Atmospheric Consequences and in This Study for Calculating Hydrospheric Consequences.

Both the RSS consequence modeling and this study employ generic sites with generic populations. Nonetheless, the results of two studies are not necessarily completely comparable due to differences in the estimation of the population-at-risk. In the RSS, that population could be more easily approximated than it could be in this study.

For the atmospheric release, most of the early health effects would be due to inhalation of part of the radioactive cloud released by the meltdown. Consequently, the population-at-risk would consist of the people downwind after the accident. External ground doses would be important at later times. The exposed population would be composed of the individuals residing in the contaminated area. Consequently, in the RSS, the dose to an individual was taken to depend primarily upon that person's location.

For the releases to the hydrosphere, most of the resultant health effects would be due to ingestion of contaminated water and foods and external ground exposure. Except for water consumption, the populations-at-risk for these pathways are not necessarily easily related to the location of the contamination. Consequently, the population-at-risk is much more difficult to delineate than it was for the atmospheric exposures. For the hydrospheric pathways, the exact doses to individuals are difficult to consider without obtaining detailed usage distribution functions for each of the pathways. Such a data gathering effort is beyond the scope of this project.

The generic sites in the RSS were defined in terms of their meteorologies. In this study, generic sites are defined according to the surface hydrologies.

1.1.6 Comparison of Atmospheric and Hydrospheric Pathways

The differences between the radiation doses received through atmospheric pathways and through hydrospheric pathways can be understood by comparing and contrasting the pathways themselves. The major differences are reviewed in Table 1.1.

1.2 Approach

To consider the potential consequences which can result by means of liquid pathways, a series of models and a set of corresponding computer programs have been assembled. The models describe the transport of the radioactivity from the site of the accident to the human population; they also estimate the resultant radiation doses and the concomitant health effects. Emphasis in this study has been placed on the development and adaptation of models for the transport of the radioactivity in the hydrosphere and in the aquatic ecosystem after accidental releases. Models previously developed for the RSS both for describing the releases of material during the meltdown process and for calculating the resultant health effects have been used here. Extensive supporting data bases have been developed for two portions of the modeling: groundwater transport and aquatic ecosystem interactions.

1.2.1 Problem

The consequences of releases into the hydrosphere after a core-melt accident would be determined by several factors: the type of release(s); the hydrological characteristics of the aquifer or other

Table 1.1 Differences Between Atmospheric and Hydrospheric Pathways

	Atmospheric Releases	Hydrospheric Releases
SOURCE	Atmosphere: primarily more volatile radionuclides (RN's) (I, Cs, Te, ...).	Melt Debris: primarily less volatile RN's (Ru, Sr, La, ...).
		Sumpwater and Depressurization: primarily more volatile RN's.
DISPERSAL	Population reached rapidly (hours).	Population usually reached slowly (months to centuries or longer).
	All RN's move essentially together; deposition mechanisms differ only for the noble gases.	Each RN moves through the ground at its own rate; each RN moves through the surface waterbodies with its own set of interactions.
PATHWAYS	Primarily inhalation and external (ground).	Primarily ingestion (drinking water, aquatic food) and external (shorelines).
	Dominant pathways are relatively simple.	Some dominant pathways are very complex.
	Populations are straightforward.	Populations are not obvious.
HEALTH EFFECTS	Acute, latent and chronic.	Primarily chronic.
INTERDICTION	Source: not possible. Pathway: possible.	Source: often possible. Pathway: possible.

substratum beneath the containment; the properties of any relatively close surface waterbodies; the potential pathways to the human population from any affected waterbodies; the interaction of the human population with any radioactivity encountered; and the possible mitigating actions to prevent consequences.

1.2.1.1 Releases to the Hydrosphere

Three types of releases directly to the groundwater could occur after a core-melt accident: leaching of the melt debris; flow of sumpwater into the ground; and injection of a portion of the containment atmosphere into the soil by depressurization. A fourth type of release to the hydrosphere could also occur: escape of sumpwater into the environment by passage to surface water along a route other than through the hole in the containment basemat formed by the melt.

In general, the first type of release could occur in any reactor for any sequence leading to meltdown;* the other three types could take place only in some reactors and only for certain accident scenarios. The first release generally would occur rather slowly; the other three, especially the last one, could take place relatively rapidly. The magnitudes of the releases and their probabilities would depend strongly upon the details of the particular accident scenario which was followed.

1.2.1.2 Transport in the Hydrosphere

As soon as the radionuclides entered the hydrosphere, dilution could take place. The amount and the rate of dilution both would vary greatly with the portion of the hydrosphere the nuclides resided in.

Initially, the radioactivity released into the hydrosphere would usually be contained in the aquifer or other stratum immediately beneath the containment. The radioactive material would be transported from beneath the reactor by groundwater to nearby bodies of surface water, such as lakes, estuaries, oceans, rivers and reservoirs. In general, this transport would take many years, although in certain situations it could take much less time. However, in some cases, the transport might be so slow that the radioactivity essentially never would reach any surface waterbody.

From the surface waterbody initially contacted, the radioactive material could be transported to other surface waterbodies. The transport in the surface water would also generally be relatively slow; residence times in specific surface waterbodies could vary from days to decades or longer.

In all of the waterbodies, the radionuclides could interact either with soil or with sediment particles. Such interactions would greatly decrease the net velocity at which certain nuclides moved in the environment. This effect would be extreme in transport through aquifers and other subsurface waterbodies. Furthermore, such interactions would drastically decrease the concentrations of some of the nuclides in the water at any point in time. Thus they would mitigate the resulting doses from certain exposure pathways. However, contaminated sediments would tend to prolong the residence period of the radioactivity in any given region. In addition, such sediments could provide alternative exposure pathways.

1.2.1.3 Consequences

The impact of an accident would be a function of the amount of radioactivity which reached the human population. This, in turn, would be dependent both upon the pathways by which the radioactivity could reach the population and upon the sizes of the populations which could be affected. The primary pathways by which human exposure could occur would be ingestion of drinking water, ingestion of aquatic foods, ingestion of irrigated foods, standing on shorelines or floodplains, and immersion in water. Any of these exposures could result in various radiation-related health effects such as cancer, thyroid nodules and genetic alterations.

1.2.1.4 Interdiction

The relative slowness with which the radioactivity would probably be carried away from the site of the accident initially often would permit time for mitigating actions to be taken. As a result, the consequences of the releases to the hydrosphere could frequently be either prevented entirely or else at least reduced substantially. Interdiction could take place either close to the source of the accident, or else farther from the site, along the various exposure pathways.

1.2.2 Methods

The description of the movement of some of the radioactivity from underneath the accident site to the human population is exceedingly complex. Computer programs have been developed to model that movement of the radioactivity, as well as to estimate the resultant consequences. The general flow of information in the computer programs is as indicated in the central column of Figure 1.2. Other programs either used to generate input data or else whose previously generated output was used as input data are shown on the left side of the figure, while the sets

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*See Appendix A for exceptions.

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of input data obtained from literature sources for this study are indicated on the right side.

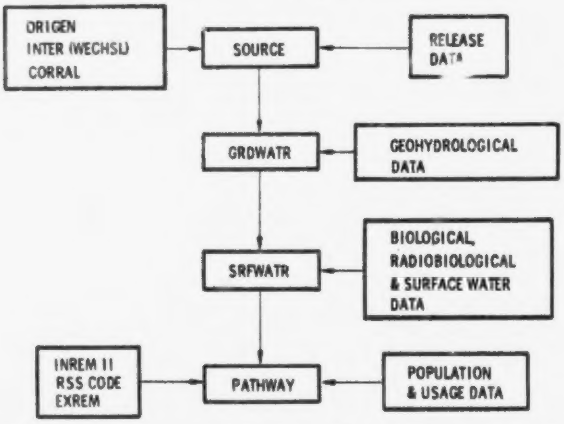

Figure 1.2 Schematic Diagram of Flow of Information in the Basic Computer Programs Used in This Study. (The general flow of information between the programs is indicated in the central column. Other programs either used to generate input data or else whose previously generated output was used as input data are given on the left side of the figure, while input data obtained from literature sources is shown on the right side.)

1.2.2.1 Releases to the Hydrosphere

Two sets of programs were used to describe the releases into the hydrosphere: (1) INTER[4] and WECHSL[5] to provide information about both the appropriate time scales for the various possible releases and the overall composition of the melt in different environments; and (2) SOURCE to calculate the amounts of radioactivity which go into each of the releases.

The interaction of the melt with the structural material of the reactor, with the concrete floor of the containment and with the soil beneath the containment were taken to be described by the models employed in INTER and in WECHSL. As is indicated in Figure 1.2, INTER and WECHSL were not part of the overall computer program developed for this study to calculate hydrospheric sequences. Instead, they were used in separate calculations to estimate the time scales over which the various types of releases might occur. Furthermore, they were used to gain insight into the chemical and physical state of the final melt in varying environments. (See Appendix A.)

SOURCE calculates the amount of every important nuclide contained in each of the three basic types of releases for any given RSS accident category and any core inventory. It also describes the time-dependent behavior of the leaching release of the radionuclides from the melt debris by utilizing any user-defined analytic expression to consider that release. The output of SOURCE consists of the amount of each radionuclide released into the groundwater as a function of time.

1.2.2.2 Transport in the Hydrosphere

Transport of the radioactivity in the hydrosphere is described by two programs: GRDWATR and SRFWATR. GRDWATR describes the transport of the radionuclides by groundwater movement from the area beneath the containment to any nearby surface waterbodies or wells; SRFWATR describes the subsequent transport in lakes, estuaries, oceans and river systems. Five types of generic sites, defined according to the nearest surface waterbody, have been dealt with: dry ("isolated" aquifer); lake; estuary; ocean; and river system. With the exception of the river system, simple analytic models have been employed to consider the movement within each waterbody; for the river system, a more complex, compartment model has been developed.

The retardation of some radionuclides caused by their interaction with the soil is included by using distribution coefficients. The analogous retardation due to the interaction with the sediment is similarly included in the surface water models. The effects of sedimentation, resuspension and diffusion are included, as appropriate, in some of the models.

In the programs, calculations are performed to determine the amount of radioactive material leaving any waterbody. Radionuclides not reaching the interface between the aquifer and the relevant surface waterbody before a specified time are eliminated from further consideration. If a relatively large fraction of the total amount of a nuclide flows from any waterbody under consideration, then that efflux of material is used as input to another waterbody, which is then taken to be the next in the series. For example, an estuary might be attached to the end of a river system.

The output of GRDWATR is the amount of each nuclide crossing the aquifer-surface waterbody interface as a function of time. The output of SRFWATR is essentially the set of concentrations of the radionuclides in the water and in the sediment as functions of both location and time.

1.2.2.3 Consequences

PATHWAY computes the radiation dose to the populations-at-risk, along with the resultant health effects. All five exposure pathways mentioned in Subsection 1.2.1.3 are considered.

Prior to computing the consequences, the program estimates the concentrations of the radionuclides in each of the exposure pathways. The radionuclides are taken to build up gradually in the relevant portions of the pathways in linear fashions according to simple models. The effects of pathway delays and processing are included in the models as indicated by the natures of the pathways.

The amount of radioactivity contained in the aquatic foods is also calculated in an alternative fashion using FOODWEB. In FOODWEB, the buildup of radioactivity in the aquatic organisms is treated using a time-dependent interacting-network model developed especially for this study. The aquatic ecosystem of each surface waterbody, other than a river, is approximated by a set of interacting physical and biological compartments. The radionuclides build up gradually in each part of the ecosystem in nonlinear fashions according to a complex set of interactions.

The health effects models in PATHWAY are essentially the same as those used in the Reactor Safety Study. Both acute and chronic effects models are utilized.

The output of PATHWAY consists of the radiation dose received by the population for each appropriate pathway as a function of each of four factors: time after the accident; accident category; type of release; and relative leaching rate. The total resultant health effects for each pathway are also tabulated.

1.2.2.4 Interdiction

The effects of mitigating actions taken both close to the site of the accident and farther along the pathways to the human population were estimated using the output of the various models already discussed.

1.2.3 Data

Two substantial data bases have been gathered for this study: (l) for groundwater transport beneath all the reactor sites in the US; and (2) for aquatic ecosystem interactions at just three particular reactor sites. The groundwater transport information was gathered from Safety Analysis Reports for most currently docketed LWR (light water reactor) sites. It was used to estimate distribution functions of some parameters used to describe various groundwater transport properties. These functions, in turn, were utilized to define a set of representative sites with respect to the subsurface hydrology. The ecosystem information was gathered from many sources for three specific sites: a large lake site; an estuary site; and an ocean site. This information was utilized to describe the time-dependent interactions within the aquatic foodwebs at each of the three sites.

No attempt was made to gather enough data to enable definition of representative sites with respect to either surface water hydrology parameters, population usage rates and sizes, or distribution functions of individual usage rates. Consequently, the specific results of the study may be limited by the values assumed and used for these properties.

1.2.4 Presentation

The material presented in this report is divided into two portions: the main body and the appendices. In general, the body contains a qualitative description of the problem; the appendices contain detailed discussions of the methods used to model the problem in this study, along with the specific parameters used in the calculations. The two treatments are usually parallel but are presented at different levels and contain different emphases.

Chapter 2 and Appendix A describe the possible direct releases to the hydrosphere in the event of a core-melt accident. The descriptions cover the entire range of time between the initiation of meltdown and the eventual release of radioactivity to the hydrosphere.

Chapter 3 and Appendix B describe the transport of the radioactive material in the hydrosphere. Detailed sensitivity studies for each of the hydrospheric transport models are included in Appendix B.

Chapter 4 and Appendix E describe the pathways by which the human population can be exposed to the radioactivity released to the hydrosphere. Appendix C considers the aquatic food pathway in greater detail. Appendix E details the potentially exposed populations for each pathway. Appendix D discusses the dosimetry and health effects models used to estimate the consequences to the human population.

Chapter 5 reviews all the models used throughout the study to consider the transport of radionuclides from the scene of a meltdown accident to the human population. It also outlines the data bases employed and the method of solution, where applicable.

The results of the study are summarized in Chapters 6 and 7. Chapter 6 contains the consequences if no mitigating procedures are employed; Chapter 7 discusses the changes in the consequences which result if various types of mitigating procedures are instituted. Appendix F discusses the types and costs of various interdictive procedures which can be employed. Appendix G contains tables of detailed results for various situations in which mitigating actions are not taken.

It is suggested that the reader who is interested only in the results of this study skip from here to

Chapter 6. The most important parts of the report are reviewed in Chapter 6 as the results are outlined. The relevant parts of Chapters 2-4 should then be consulted if more insight into any portion of the overall problem is desired. The information in those chapters should also be reviewed if an appreciation of the factors affecting the uncertainties in the results of this study are important. In contrast, Chapter 5 and Appendices A-E should be referred to if a greater understanding of any of the models and/or the related data used in this study is the goal. That information should also be consulted if knowledge about the sensitivities of the results (due to the properties of the specific models utilized) is the aim.

References

[1] U. S. Nuclear Regulatory Commission, "Reactor Safety Study," WASH--1400, NUREG-75/014, USNRC, 1975.

[2] Offshore Power Systems, "OPS Liquid Pathway Generic Study," OPS, 1977.

[3] U. S. Nuclear Regulatory Commission, "Liquid Pathway Genetic Study," NUREG-0440, USNRC, 1978.

[4] W. B. Murfin, "A Preliminary Model for Core/Concrete Interactions," SAND77-0270, Sandia Laboratories 1977.

[5] H. Alsmeyer, W. Murfin, and M. Reimann, "Hydrodynamische und thermische Modelle zur Wechselirkung einer Kernschmelzen mit Beton," *PNS Halbjahresbericht 1978/1*, KfK2700, Kernforschungszentrum Karlsruhe, GmbH, September 1978.

2 Description of Melt-Through

2.1 Introduction

This chapter contains a description of those characteristics of a meltdown which are important to liquid pathway considerations. In Sections 2.2 and 2.3, the possible types of releases to the hydrosphere are described. In Section 2.4, the dependence of those releases upon the type of reactor is reviewed. In Sections 2.5 and 2.6, the probabilities and compositions of the possible releases to the hydrosphere are considered. The details of related calculations and explanations are presented in Appendix A. It should be noted that the same reference reactors that were discussed in the Reactor Safety Study[1-3] are considered in this report.*

2.2 Types of Releases

2.2.1 Leaching of Melt Debris

Melting of a reactor core could occur if the heat generation rate exceeded the rate of heat removal. After the initiation of such liquefaction, structural materials from the reactor vessel would be gradually incorporated into the molten core. Eventually the heat-generating mass would melt through the bottom of the reactor vessel and would fall onto the concrete of the containment floor, where it would interact both chemically and physically with the concrete. The decay heat of the melt would decompose and/or liquefy the concrete, so that the melt would continue to migrate downward. As the molten mass moved through the concrete, it would increase in size by accreting the nongaseous decomposition products of the concrete. If it descended entirely through the containment floor, it could continue to grow by incorporating the "soil" underneath the containment. This agglomeration would continue until the decay heat of the radioactive core had been diluted sufficiently by accretion and had been decreased adequately by decay that further melting and decomposition of the surrounding media did not occur.

If complete melt-through of the containment floor occurred, then after sufficient cooling of the mass, groundwater could contact it and begin to leach out the various radionuclides. Substantial groundwater contact with the melt would probably be delayed for some time after melt-through due to the formation of a dry area around the melt debris. In all likelihood, permanent contact with the groundwater would take several months to years to be initiated, with the time depending primarily on the availability of water in the subsurface region.

The absolute rate at which the nuclides would be leached would depend upon both the physical and the chemical characteristics of the final mass. The overall chemical character of the melt would be determined, in turn, by the composition of both the basemat concrete and the underlying soil. Because concretes of widely varying compositions are used in constructing reactor containments and because almost all categories of soil and rock types are found beneath existing reactor containments, the possible final composition of the melt could vary over a broad spectrum. Inasmuch as leaching rates are strong functions of the composition of the substance being leached, there would be, at the least, a rather wide range of leach rates possible for any given element due to the essentially infinite variety of possible compositions of the final melt.

The overall physical character of the melt would be determined by both the accident scenario and the relevant properties of the substrate. Due to the continuous cooling of the mass by gaseous convection, the residual mass might be quite porous. Furthermore, cooling could result in substantial fracturing of the mass. Thus, once groundwater contacted the mass, it could probably contact a significant surface area. As a result, the leaching might not be significantly limited by the apparent surface area.

The rates of leaching would depend not only on the character of the melt but would depend also upon the groundwater flow in the region. Leaching could generally proceed more rapidly in regions of higher flow rates. After adequate groundwater flow were established, leaching of the radionuclides from the debris could take from weeks to very many years.

2.2.2 Depressurization of Containment Atmosphere

If no rupture of the containment occurred prior to melt-through of the basemat, then part of the containment atmosphere could be driven into the ground beneath the reactor through the hole formed by the melt. The material in this depressurization release would include both gases and particulate matter. This material could be expected either to migrate to the surface in gaseous forms or to go into solution in any groundwater which might be present around the containment basemat. Inasmuch as the material would be in a highly dispersed form, any soluble components would be expected to go into solution rather rapidly. In contrast, insoluble gases, such as the noble gases, would tend to rise to the surface rather than go into solution.

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*In the RSS, two specific reactors, a pressurized water reactor (PWR)[1] and a boiling water reactor (BWR),[2] were used to consider all the in-reactor aspects of a meltdown accident. Thus, some parts of the discussion of a meltdown in a "generic" reactor given in both that study and in this one are biased by their consideration of only these two reactors.

The time which any of the insoluble gaseous components would take to percolate to the surface would depend upon the conditions of the "soil" beneath and around the containment. In porous sandy soils the migration to the surface might take just hours while in much less permeable soils the movement could take months. Any structural modifications of the strata under and around the containment made during construction could impact significantly on the time it would take any gases to migrate to the surface.

2.2.3 Sumpwater Escape

In addition to the movement of the melt into the substratum beneath the containment and the depressurization of the containment atmosphere into the soil, a third type of direct release of radionuclides into the soil could occur. Specifically, all or part of the sumpwater could drain through the hole in the containment floor formed by the passage of the melt. If the containment sprays operated during the melt-through, then a substantial portion of the radionuclides initially released into the containment atmosphere could be found in the sumpwater.

If the containment failed prior to melt-through, water contacting the hot melt could vaporize into steam and be lost from the containment. Such vaporization could hasten cooling of the melt so that earlier permanent contact between the groundwater and the melt might be established. As a result, it might allow leaching of at least part of the melt to start at an earlier time than would otherwise be expected. In contrast, if an alternate path around the melt debris existed, the sumpwater might drain into the ground without evaporation. Such a path might be provided by cracks in the basemat formed by the stresses and strains of the melt-through.

The time that it would take for the sumpwater to drain into the subsoil would be determined by the relative position of the groundwater table at the containment and by the hydrogeological characteristics of the soil. As in the case of a depressurization release, the two most relevant hydrogeological parameters would be the permeability and the effective porosity of the soil. If the sumpwater drained under its own head, which it probably would, then it could take from days to years for the drainage to be completed. In view of the fact that the radionuclides in the sumpwater would already be in solution or at least in suspension, their entry into the groundwater system would be limited primarily by the drainage rate of the sumpwater.

The sumpwater might also enter the hydrosphere by an alternative route to the one just considered: it might be released directly to accessible portions of the environment by some pathway other than through the hole in the basemat formed by the melt. The rate of release would be dependent on the specific pathways which were followed by the water.

2.3 Movement into the Hydrosphere

Any of the three releases which left the containment by transit through the hole formed by the melt would enter the hydrosphere via movement into the groundwater. If the melt came to rest above the local watertable, then the entrance of the radionuclides into the hydrosphere by leaching of the melt debris would be substantially delayed. However, the melt debris for most LWR's (light water reactors) would reside below the watertable (see Appendix B). Therefore, as soon as sufficient cooling had taken place, the radionuclides could be leached from the debris.

Likewise, if the containment floor were above the watertable, then the gases found in the depressurization release might just flow upward to the surface rather than being entrained by the groundwater. Only if the containment floor were beneath the watertable would a substantial fraction of the depressurization release be entrained.

In contrast, even if the containment floor were located above the watertable, sumpwater could percolate down to the groundwater. In fact, such a situation might be advantageous for the entry of sumpwater because it would indicate less resistance to the flow of the sumpwater from the containment.

2.4 Comparison of PWR and BWR Melt-Through

2.4.1 Reference Reactors

Since the containment and the reactor designs are quite different for PWR's (pressurized water reactors) and for BWR's (boiling water reactors), the actual

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pathways followed by the melts in each of these types of reactors could be rather dissimilar. In a reactor such as the Reference PWR (see Figure A2 in Appendix A), it is expected that any melt-through would inevitably occur by passage through the reactor cavity floor. In contrast, in the Reference BWR (see Figure A4 in Appendix A), the melt might follow one of at least two options. As in the PWR, the core could also melt its way through the reactor cavity bottom. Alternatively, the melt in a BWR could breach the side walls of the cavity. Such support-wall breaching could result in the retention of the melt within the BWR containment.

For complete melt-through of the Reference BWR, the basic parameters describing the final melt -- the time required to penetrate through the containment floor, the final volume and the composition of the melt, and the cooling time -- would all be expected to be similar to those of the Reference PWR. However, for shield-wall breaching, the characteristics of the final melt could differ substantially from those expected for complete melt-through of the containment basemat.

Not only could the paths taken by the melt differ in a PWR and in a BWR, but the possible radioactive release to the hydrosphere could also be different for the two types of reactors. In addition to the release of the melt itself into the groundwater, both sumpwater and depressurization releases could occur for a Reference PWR;[1] however, for a Reference BWR,[2] neither of these last two releases would be possible. For such a BWR, there are no containment sprays and rupture of the containment would always occur prior to melt-through.

2.4.2 Other Reactors

The probabilities and the magnitudes of the various radioactive releases which might occur at a given site would both be extremely dependent on certain design features of the particular reactor under scrutiny. Reactors other than the "reference" ones dealt with in the RSS and in this study differ substantially in some of these features (see Section A6 in Appendix A).

2.5 Characteristics of Releases

2.5.1 Probabilities and Relative Magnitudes

The probabilities of various possible meltdown sequences have been estimated previously in the RSS.[3] Using these values, the probability for each of the possible releases to the hydrosphere through the hole formed by the melt can be estimated as follows: the probability for complete melt-through of the containment basemat is approximately that for meltdown;* the probability of sumpwater release (from a PWR) through the melt hole is that of meltdown with both shield-wall failure and sufficient sumpwater availability;** the probability of partial depressurization (the containment sprays work so that the containment pressure is relatively low) is approximately that of meltdown with initial rupture of the containment by melt-through; and the probability of complete depressurization (the containment sprays do not work so that the containment pressure is rather high) is given by that of initial containment rupture by melt-through concurrent with failure of the containment sprays (see Table 2.1).

Table 2.1 Probabilities of Release (per reactor-year)

Event	Probability
Melt-through	4×10^{-5}
Depressurization	
PWR-complete	6×10^{-7}
PWR-partial	1×10^{-5}
BWR	0
Sumpwater Escape	
PWR	5×10^{-6}
BWR	0

The types and magnitudes of the releases which would occur during a meltdown would be a function of the particular accident scenario which were followed. In the most probable PWR meltdown accident category (PWR-7[3]), the containment sprays would operate and the containment would not rupture except by the melt-through itself. Thus, the sumpwater might contain essentially all the radioactivity initially released to the containment atmosphere, except for the noble gases. In this accident category, the activity in the containment atmosphere, and therefore in the sumpwater, would be at its maximum possible value for all categories. Because no steam explosions occur in this most probable category, the amount of radioactive material in the melt debris would also be at its maximum value. Consequently, the most probable

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*It is possible for the melt to be permanently coolable either in the reactor vessel or in the reactor cavity so that melt-through of the basemat would not necessarily occur for every meltdown. See Subsection A4.3 of Appendix A.

**The probability of sumpwater release by routes other than through the hole formed by the melt is not included here.

PWR meltdown category could result in almost all the available radioactivity being released to the hydrosphere and not to the atmosphere (see Table 2.2 in this chapter and Tables A14 and A15 in Appendix A). Most of the accident categories would result in approximately the same amounts of the dominant radionuclides being contained in the melt debris for a PWR meltdown (Table A14). However, the amounts contained in the other types of possible PWR releases would be much more dependent upon the accident category (see Tables A11 and A15 in Appendix A).

The only major release to the hydrosphere for a BWR meltdown is expected to be by leaching of the melt debris. The amount of radioactivity in the debris would be approximately the same for all BWR accident categories except the one in which significant steam explosions occur. However, that category is the least probable one. As a result, essentially the same amount of radioactivity is expected to be contained in the melt debris in most BWR meltdowns (Table A14).

Table 2.2 Best Estimates for the Fractions of the Initial Radionuclide Inventory Entering Various Releases for the Most Probable PWR Accident Category(a)

Element Classes	Release Fractions Leach	Sumpwater
I	0.00	1.00
Cs(b)	0.00	1.00
Te(c)	0.00	1.00
Sr(d)	0.89	0.11
Ru(e)	0.92	0.08
La(f)	0.99	0.01

[a] The best estimates of the release fractions for the most probable accident category (PWR-7) are representative of those for the majority of accident categories.

[b] Includes Rb.

[c] Includes Sb.

[d] Includes Ba.

[e] Includes Ru, Rh, Co, Mo and Tc.

[f] Includes Y, La, Zr, Nb, Ce, Pr, Nd, Np, Pu, Am, and Cm.

2.5.2 Radioactive Composition

The radioactive content of the releases would be determined essentially by the inventory of radioactive fission products, actinides and activated structural materials present in the core at the time meltdown were initiated. The volatile components of the melt would be released initially to the containment atmosphere. From there, these species would be released either to the atmosphere outside the containment (if the containment were ruptured by other than melt-through), to the sumpwater (if the containment sprays worked), or to the ground by depressurization (if the initial rupture of the containment were via melt-through). The less volatile components would tend to be retained in the melt itself.

The volatile core components include the halogens such as iodine, the chalcogens such as tellurium, and the alkali metals such as cesium (Table 2.2). The relatively non-volatile core components include the lanthanides such as neodymium, the actinides such as plutonium, and the alkaline earths such as strontium (Table 2.2).

With the notable exception of cesium, the dominant volatile radionuclides present in any releases to the environment generally would be relatively short-lived. Within five years after the accident, the activity of the initially volatilized components would be essentially just that of the cesium (Figure 2.1). Most of the longer-lived isotopes present in significant quantities in the core at the time of meltdown would be comparatively non-volatile. Consequently, they would be found primarily in the melt debris. Within five years after the accident, essentially half of the activity of the initially non-volatilized species would be due to strontium (Figure 2.1). At longer times, the activity would be partially due to the actinides and their decay chains and partially due to the lanthanides. After much longer times, the actinide chains would dominate the remaining activity in the debris.

It should be noted that both the melt debris and the sumpwater could contain substantial fractions of all the radioactivity available for release in a core-melt accident. In contrast, the depressurization release would typically possess only a small portion of that total radioactivity (see Table A11 in Appendix A).

2.5.3 Nonradioactive Composition

The volatilized species would include both radionuclides from the core and some non-radioactive species such as certain structural materials released from the reactor and various gases generated in the decomposition of the concrete. The non-radioactive gases present in the containment atmosphere would not be expected to significantly affect the availability of the radioactive gases for release either by sumpwater escape or by depressurization, other than primarily by their effect on the containment pressure. In addition, the presence of certain dissolved species in the sumpwater (spray water) could affect the solubility of others.

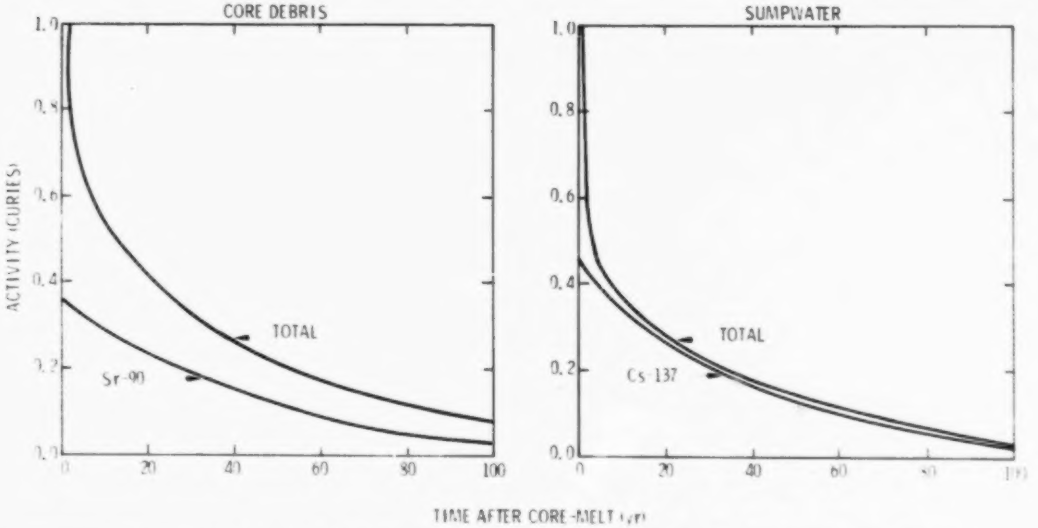

Figure 2.1 Time-dependent Behaviors of the Radioactive Constituents of the Melt Debris and the Sumpwater.

The non-volatilized species residing in the melt debris would include parts of the core, some structural materials and concrete and soil residua. Because the majority of the melt would be composed of non-core components, the non-radioactive content of the melt would be very important for determining the rate of release to the hydrosphere by leaching of the melt debris. (See Section 2.6 and Appendix A.)

2.5.4 Time Scales

The amount of radioactivity which could reach the human population from any of the releases to the hydrosphere would depend on both the rates of release and the rates of decay of the dominant radionuclides. The depressurization and the sumpwater releases into the ground would generally be relatively rapid releases (Table 2.3). In view of the fact that most of the radionuclides present in significant quantities in those releases would be relatively short-lived, any substantial delay in either of these releases could significantly reduce the potential exposure by the human population.

In contrast to the other two types of releases, the leach release would typically be relatively slow (Table 2.3). Because most of the long-lived radionuclides present in significant quantities in releases to the hydrosphere would be found in the leach release, the relative reduction in the potential population exposure caused by any delays of the leaching would not be as large as for any comparable delays of the other two types of releases.

Table 2.3 Times for Various Melt-Through Related Processes[a]

Time Period	Time
Melt-Through	
Time to Contact Concrete	
PWR	2 to 28 hr
BWR	3 to 33 hr
Time to Melt-through[b]	
PWR SiO_2 concrete	72 hr
$CaCO_3$ concrete	72 hr
BWR SiO_2 concrete	72 hr
$CaCO_3$ concrete	72 hr
Releases	
Time for Depressurization Release[c]	
PWR	hr to yr
Time for Sumpwater Release[c]	
PWR	hr to yr
Time to Groundwater Contact of Melt Debris[c]	
PWR and BWR	mo to yr
Time for Leaching of Melt Debris[c]	
PWR and BWR	wk to yr
Transport	
Time to Reach Humans	
PWR and BWR	hr to ∞, usually at least yr[d]

[a] See Appendix A for details.

[b] Time after concrete contact.

[c] Time after melt-through.

[d] See Chapter 3 and Appendix B for details.

Obviously, the important factor in determining the potential effect of a given radionuclide for the leach release would be whether its leach rate were large compared to its decay rate. Only if significant amounts of a given radionuclide were leached prior to decay could it be important.

Any release of radioactivity into the hydrosphere after a meltdown would generally be directly into the groundwater. After any of the three releases had entered the groundwater, transport to an accessible portion of the hydrosphere would need to occur before human exposure could take place. Such a location could be either a well or a surface waterbody. This transport could take from less than a day to an essentially infinite amount of time. In general, it would take at least years. (This transport is discussed in the following chapter.)

In contrast to the other three types of releases to the hydrosphere, the direct release of sumpwater into surface water could be very rapid and might take just hours. Thus, human exposure could occur very soon after the accident.

2.6 Nonradioactive Composition of Melt

The nonradioactive composition and character of the melt would ultimately determine the rates and even the magnitudes of the releases of radioactivity to the hydrosphere, especially for the release by leaching of the melt debris. Among the primary determinants of those properties of the melt would be the accident sequence followed and the types of soil and concrete contacted.

2.6.1 Accident Sequence

Before the melt contacted the concrete of the containment basemat, the melt's properties would be determined by the dominant chemical and physical reactions taking place during the early stages of the accident. These could be described roughly by considering two factors: the time from accident initiation to contact with the concrete; and the amount of water available to the melt. Both these two factors would be functions of the relevant accident sequence.

If the time from accident initiation to contact of the melt with the basemat concrete were comparatively long, substantial amounts of the structural materials could be incorporated into the melt. If water were available, many of the metals involved in the melt could be oxidized early in the meltdown process. In addition, the presence of water could permit a steam explosion to occur; as a result, a significant portion of the core material might be scattered from the melt, making melt-through of the containment basemat less likely. Furthermore, water could be important for cooling the melt and delaying its progress.

Once the melt contacted the concrete, a third factor could affect the composition of the melt: the pressure inside the containment. This pressure would be important in describing any accident sequence insofar as any of the reactions which occurred were pressure dependent. For example, for high containment pressures, the decomposition of $CaCO_3$-based concrete would be somewhat self-limiting as long as the containment were intact.* As a result, high pressures could increase the melt-through time in such concretes.

2.6.2 Concrete and Soil

In general, the materials incorporated into the melt from the concrete and the soil would comprise at least 85 to 95% of the final melt's mass, with core and structural materials forming the complement of that mass. The concrete and the soil beneath the containment would determine not only the overall composition of the melt but also the final configuration of that mass. The reactions undergone by the dominant constituents of the concrete and the soil with the melt would be largely responsible for that configuration.

For example, the predicted configuration of the final mass for a meltdown involving a silicon dioxide-based concrete basemat overlying a silica substrate (sand) is shown in Part A of Figure 2.2 and that for a calcium carbonate-based concrete base mat overlying a carbonate substrate (limestone) is shown in Part B of that figure. The silicon-based concrete and substrate and the carbonate-based concrete and substrate can be considered to represent the range of possibility of the most important aspects of concrete and soil conditions. The primary reaction of silicon dioxide to the decay heat would be melting; the reaction of the carbonate would consist of an initial decomposition, followed by a melting of the residual-calcium oxide. The heat required to melt silicon dioxide is approximately an order of magnitude less than that required to decompose and to melt an equivalent amount of calcium carbonate. Therefore melts in the two types of materials would probably vary greatly with respect to overall size and depth of penetration. As expected, the melt would probably not travel as far into the soil in the limestone substrate (Figure 2.2).

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*Any chemical reaction which involved a net change in the amount of gases present in the containment atmosphere would be very pressure dependent. The decomposition of calcium carbonate ($CaCO_3$) results in the evolution of carbon dioxide (CO_2) gas. Inasmuch as the equilibrium state of any chemical reaction tends to a position which reduces any stress applied to the reaction, $CaCO_3$ decomposition will be suppressed by high CO_2 pressures.

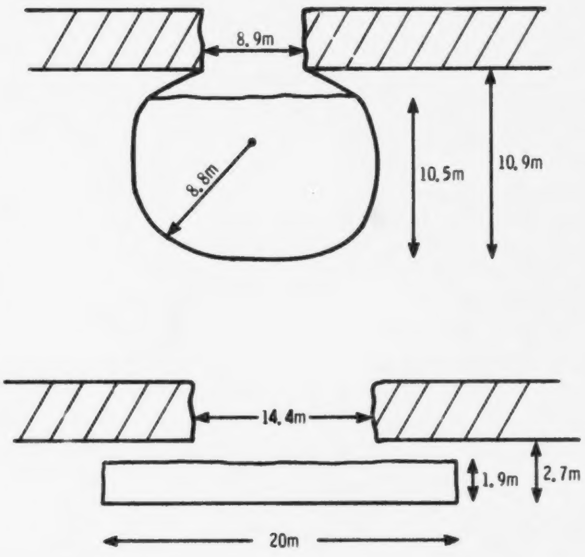

Figure 2.2 Final Configuration of the Melt Debris for Melt-through into (a) Siliceous Concrete Overlying Siliceous gravel; (b) Limestone-based Concrete Overlying Limestone Gravel.

The heights of the melts in the holes shown in Figure 2.2 are for representative situations. In reality, these heights could be either higher or lower than those indicated. If a melt did not progress as far into the substrate as either of the melts depicted in Figure 2.2, so that part of the melt blocked the hole formed by its passage, sumpwater and depressurization releases would still probably be possible. This is because the movement of fluids through the hole might not be significantly impeded as a result of the porous and fractured character of the melt debris.

The masses which would be formed by each of these two types of concrete and soil conditions also might represent possible extremes with respect to leaching behavior. Concrete and soil high in calcium would form a melt which had the basic composition of a calcine. Such materials generally have relatively high leach rates for most elements. In contrast, concrete and soil high in silicon would form a mass which had the chemical composition of a glass but which might have much different physical characteristics. Thus, leaching from such a mass would probably be lower than anticipated from a calcine but much higher than would be expected from a carefully-prepared nonporous glass. Most concrete and soil compositions in general would fall somewhere between the extreme calcium and silicon cases just considered.

2.7 Summary

Three types of releases to the hydrosphere are possible as a result of the escape of radionuclides through the hole formed in the containment basemat by the melt: leaching of the melt debris; entrainment of escaping sumpwater; and dissolution of depressurized containment gases. The first type of these releases probably can occur for any meltdown accident while the last two can occur only for meltdowns at certain types of reactors. In addition to release through the melt hole, sumpwater can be released to the hydrosphere by other pathways from the containment. Such alternate routes may exist at many reactors for which sumpwater release is possible.

The leaching of the melt debris would constitute the slowest release to the hydrosphere. Initially, this release would probably be delayed for a period of months to years by failure of the groundwater to continuously contact the debris. The leaching period could be expected to range from months to many years, depending upon the composition of the debris and upon the radionuclides under consideration.

The depressurization release and the sumpwater release via the melt hole would generally be more rapid than the leaching of the debris and could take from hours to years, depending upon the subsurface conditions at the site. In contrast, if sumpwater were released directly to the surface water, then that release could conceivably take just hours. Such a release normally would reach accessible portions of the environment many years before any of the three releases into the ground would.

The most important parameters influencing the chemical and physical form of the melt would be the type of concrete in the basemat and the composition and water content of the underlying soil. In any melt, the core and reactor materials would be highly diluted and would represent only a very small fraction of the total final mass. The melt could probably be characterized as porous and highly fractured.

Both the melt debris and the sumpwater could potentially contain substantial fractions of the total radionuclide content of the reactor prior to meltdown. In contrast, any depressurization release would generally possess only a comparatively small fraction of the total radioactivity.

The dominant category of PWR core-melt accidents is the one with both the largest leach release and the largest potential sumpwater release. Almost all the core-melt accident categories are predicted to have essentially the same amount of activity in the melt debris.

Except for cesium, the long-lived isotopes present in substantial quantities in the releases to the hydrosphere would be contained primarily in the melt debris. In contrast, the sumpwater and the depressurization gases would contain significant quantities of mostly short-lived isotopes, with the exception of cesium.

Both the magnitudes and the probabilities of any releases to the hydrosphere would be extremely site dependent. Among the important factors would be the following: the type, the size and the status of the reactor; the design characteristics of the containment (see Section A6 in Appendix A); any construction beneath the containment (see Subsection F2.2 in Appendix F), and the type of concrete in the containment basemat and the type of soil beneath it.

References

[1] Virginia Electric and Power Company, *Surry Power Station Units 1 and 2, Final Safety Analysis Report*, VEPCO, 1969.

[2] Philadelphia Electric Company, *Final Safety Analysis Report, Peach Bottom Atomic Power Station Units No. 2 and 3*, Philadelphia Electric Company, 1972.

[3] U.S. Nuclear Regulatory Commission, *Reactor Safety Study* (WASH-1400), NUREG-75/014, USNRC, 1975.

3 Groundwater and Surface Water Transport and Dispersion

3.1 Introduction

As soon as the radionuclides entered the hydrosphere, continuing processes of transport and dilution would be initiated. The factors which would contribute to this transport and dilution are discussed in this chapter.

In this study, five major types of generic reactor sites, differing only in their surface hydrologies, are considered: "dry," large lake, estuary, ocean, and river system. A "dry" site is taken to be one with no relatively close bodies of surface water; that is, it is a site at which contamination of only underground water could probably occur. In contrast, at each of the four surface-water sites, contamination of both groundwater and surface water could occur.

This classification of site types is somewhat arbitrary. Obviously, each of the five types can be subdivided. Furthermore, each of these waterbodies is not isolated but, rather, is connected to other waterbodies. As a result, an accidental release into one waterbody could result in significant impacts in successive waterbodies.

In this chapter, the relevant features of the hydrologies of the five categories are considered. In Section 3.2, the general factors which affect movement of the radioactivity through the hydrosphere are discussed. In the next two sections, 3.3 and 3.4, the specific factors which determine movement through the five considered types of waterbodies are reviewed. In Section 3.5, the hydrospheric factors most strongly affecting the resultant consequences of a meltdown accident are summarized. The details of related calculations and explanations are presented in Appendix B.

3.2 General Considerations

3.2.1 Hydrospheric Transport

Movements of water and any associated radionuclides within the hydrosphere can be divided into two types: transport and dispersion. The former classification consists of those processes which result in average net movement of masses of water. Such transport movement includes both advective and convective processes. The latter, dispersive, classification of movement types consists of those processes which result in non-average behavior of specific particles. In considering the movement of radionuclides through the hydrosphere, dispersion describes the spreading of the radionuclide mass relative to the movement of its center of mass.

The type, the size and the status of the waterbody in which a radionuclide mass finds itself determine which kind of movement predominates the behavior of the mass. If the amount of dispersion is not significant, then three factors generally determine the resulting concentraion of a radionuclide in any waterbody: the rate at which the radionuclide is released into the waterbody; the rate at which the material is initially diluted when it enters the body; and the rate of removal of the radionuclide from the waterbody by processes such as flushing and decay.

3.2.2 Sediment and Soil Behavior

3.2.2.1 Behavior

The primary effect of soil and sediment on the transport of radioactivity in the hydrosphere is due to the tendency of soil and sediment particles to adsorb or otherwise interact with some of the radionuclides. As a result of this tendency, both soil and sediment particles can temporarily, or in some cases permanently, remove significant quantities of radionuclides from the water. The resultant radionuclide concentrations in the soil or sediment can even be much greater than the concentrations present in the associated water.

In the soil, the net effect of the interactions is to substantially delay the movement of many of the radionuclides in the soil relative to the movement of the water. In a surface waterbody, the net effect of the interactions depends primarily upon the fate of the sediment. In general, the effect at short times is to lower the concentration of the radioactivity in the water and at longer times is often to act as a secondary source of radioactive contamination for the water. As a result, the movement of some of the radionuclides in surface waterbodies can effectively be somewhat delayed because of sediment-nuclide interactions.

3.2.2.2 Differences Between Soil and Sediment

The primary differences in the interactions and their effects in soil and in sediment are due to two factors: the soil is fixed in space whereas the sediment is relatively free to move; and, the proportion of soil to water is much larger than the ratio of sediment to water. An important distinction which results from the differing mobilities is that a large portion of a radionuclide mass can be confined to a relatively limited region of soil whereas it might not be as confined in the sediment because the sediment can

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move. In particular, after nuclides are attached to the sediments, they can be moved substantial distances rather rapidly by sediment movements. The primary effect of the larger ratio of soil to water than sediment to water is that proportionally much more of the total mass of a nuclide can reside on the soil than on the sediment. This is because of the much larger interaction surface presented by the soil. Thus the soil can be substantially more effective both in lowering the radionuclide concentration in the water and in retarding the movement of the radionuclide than the sediment can be.

In surface waterbodies, the interactions can often be regarded as causing perturbations of the more fundamental movement of the radionuclides by the water itself. In aquifers, the interactions must be regarded as being much more than mere perturbations.

3.2.3 Soil-Nuclide and Sediment-Nuclide Interactions

3.2.3.1 Interactions

The chemical state of an element is crucial in determining the interactions that element will undergo with its environment. The important factors for determining that chemical state can vary strongly as functions of both space and time. Consequently, the tendency of movement of a given species can change quite rapidly. This is particularly true at various interfaces such as the sediment-water interface or the interface between two distinct waterbodies such as an aquifer and a lake.

The amount of interaction between the soil or the sediment and any nuclides is also a strong function of the composition of the soil or the sediments. Therefore, the important interactions can vary dramatically as the character of the soil or sediment changes.

3.2.3.2 Effects of Interactions

Elements characterized as being relatively reactive in the hydrosphere tend to become associated with either soil or sediments (or biological organisms) and are thus effectively removed from solution. Because of their association with soil or sediments (or organisms), they are essentially localized in the vicinity of their interaction unless transport of the interaction medium (either the soil or sediment or organisms) occurs. As a result, their residence times in an area generally are expected to be much longer than the residence time of the water.

Elements characterized as being relatively unreactive in the hydrosphere tend to remain in solution rather than becoming associated with soil or sediments (or organisms). Consequently, they tend to move with the mass of water. Their residence times in an area are expected to be approximately the equal to residence time of the water itself.

3.3 Groundwater Movement

In any liquid pathway, the initial movement of a radionuclide away from the scene of the core melt would usually be by groundwater transport. Consequently, the processes which effect the movement of any radionuclide through the ground are considered here.

3.3.1 Water Movement

If the groundwater at the site of the melt-through were higher than the final resting place of the melt, the radionuclides could be gradually leached out of the debris into the groundwater. Once entrained in the groundwater, the radioactivity could be transported through the soil to any surface waterbody in the proximity. The rate of transport would depend upon the structure and composition of the soil through which movement had to occur.

The stratigraphy beneath the containment at the site of a core-melt accident could be such that either an aquifer, an aquitard or an aquiclude could contain the final melt. An aquifer bears water relatively easily; an aquiclude stores water but does not transmit significant amounts of it; and an aquifuge effectively neither transmits nor stores water. The water-bearing properties of the actual stratum or strata beneath the containment would be very important in determining the rate at which radioactivity could be moved. Rates of flow in aquifers can vary from less than a meter per year to more than a hundred meters per year. The rates of flow in aquitards and in aquicludes are much less than those in aquifers. Only strata classifed as aquifers typically would permit substantial movement of radioactivity over relatively short time scales (hundreds of years or less).

Reactors are situated at distances of less than one hundred meters to more than ten thousand meters from the nearest significant surface waterbodies (see Figure 3.1). The time it takes for water to flow from beneath an LWR containment to the nearest waterbody (the groundwater travel time) depends upon both the distance to the water and the rate of groundwater flow at the site. Estimates of this time vary from less than a day to more than thousands of years (see Figure 3.1 and Appendix B).

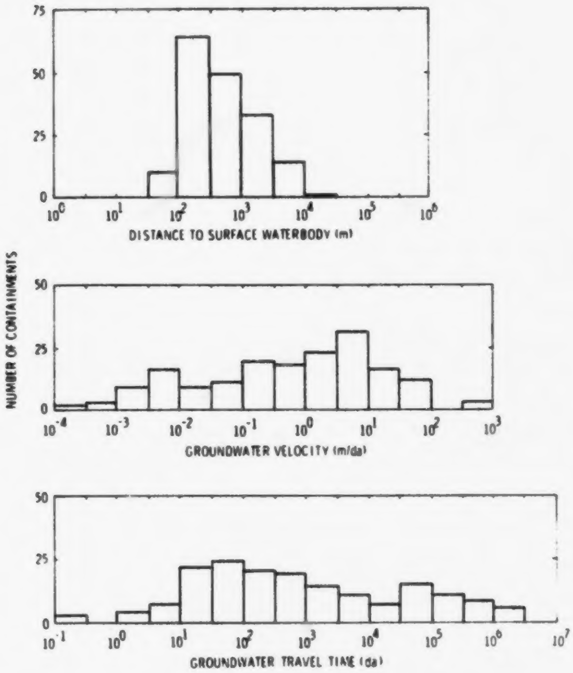

Figure 3.1 Histograms of the Distances of the LWR Containments from the Nearest Surface Waterbodies, the Groundwater Velocities at the Reactor Sites, and the Average Times Groundwaters Take to the Nearest Surface Waterbodies.

Upon the entry of any radionuclides into the groundwater, several processes would tend to cause dispersal of the material beyond that caused by the flow of the groundwater. The net effect of the dispersal would be to spread the mass of radioactivity over a much wider area than would otherwise be expected. Consequently, even an initially instantaneous release into an aquifer would reach any surface waterbody or well over a rather extended period of time. Part of the material would reach the waterbody some time before the bulk of the material and part would reach it some time afterwords. In very irregular substrates, some of the radioactivity could reach accessible portions of the hydrosphere long before the bulk of the material. Therefore, significant exposure might occur much earlier than would otherwise be expected.

Dilution of any material would occur much more slowly in an aquifer (or other subsurface waterbody) than in any surface waterbody. As a result, the material in the aquifer could remain rather concentrated over long periods of time. The concentration in an aquifer would be limited primarily by the initial rate of entry of material into the substratum. Therefore, the largest exposure of an individual via the hydrosphere could occur by direct access to contaminated underground water.

3.3.2 Soil-Nuclide Interactions

The delay in the movement of some radionuclides caused by their interactions with the soil particles could be substantial (for example, see Figure 3.2). The retardation of a nuclide relative to the water carrying it could even be up to several orders of magnitude. As a result, the dispersion caused by this retardation could also be significant (also see Figure 3.2).

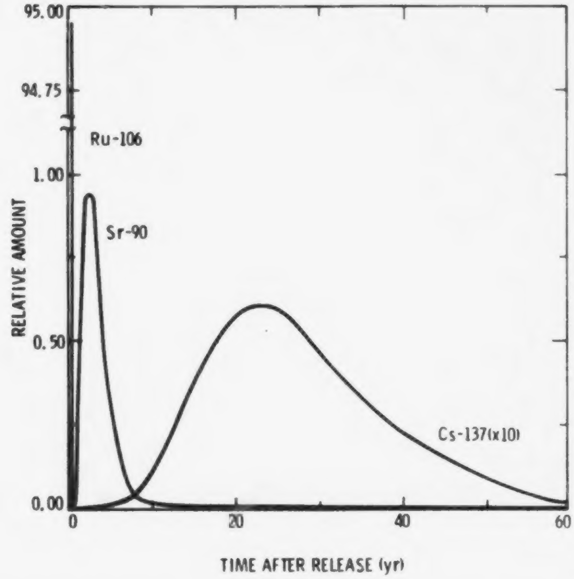

Figure 3.2 Relative Amount of Radioactivity Crossing Interface After a Unit Release of Each Radionuclide. (The groundwater travel time to the interface is 10 days; the distribution coefficients are 0, 20, and 200 for Ru-106, Sr-90, and Cs-137, respectively.)

The average relative velocity of any element with respect to the velocity of groundwater can be obtained from knowledge of its distribution coefficient, K_d. (This coefficient is conventionally defined for a given radionuclide as the ratio of the concentration of radionuclide adsorbed on the soil (or sediment) to the concentration of radionuclide in solution). The average speed of any element is approximately inversely proportional to its distribution coefficient. Thus a large distribution coefficient indicates a relatively slow velocity for the element (see Figure 3.2).

Inasmuch as the distribution coefficients of the radionuclide would differ by orders of magnitude, the various radionuclides would enter the surface water at a given site at vastly different times (also see Figure 3.2).

Because the average speed of any radionuclide would be proportional to the speed of the groundwater, the corresponding radionuclide travel time to the nearest surface waterbody would be proportional to the groundwater travel time to that body. Inasmuch as the groundwater travel times for LWR sites vary over orders of magnitude (see Figure 3.1), the radionuclide travel times for a given radionuclide would likewise vary over orders of magnitude. As a result, even small differences in the groundwater travel times could correspond to large differences in the amounts of some of the radionuclides which would cross the groundwater-surfacewater interface before decay.*

In general, the dominant components of the melt debris would have relatively large distribution coefficients whereas the dominant ones of the sumpwater and depressurization releases would have relatively small distribution coefficients. Furthermore, those species which would be leached more rapidly from the melt debris would tend to have smaller distribution coefficients than the majority of the components of the debris (see Table A12 in Appendix A and Table B3 in Appendix B; also see Figure 3.3). Consequently, the most rapidly released components would usually be those which could be expected to travel most rapidly through the soil. The notable exceptions to these generalizations would be ruthenium and strontium for the melt debris, and cesium for the other two releases.

3.4 Surface Water Transport

At four out of five of the generic sites, movement through a surface waterbody would play a significant role in the dispersal of radioactivity through the environment. Consequently, the physical movement of radionuclides in such waterbodies is now considered.

After the radioactivity entered any surface waterbody, it would be diluted and transported at a much faster rate than it would be in the ground. The rate of movement through and within the surface waterbody would depend primarily upon the general advective and convective characteristics of that waterbody.

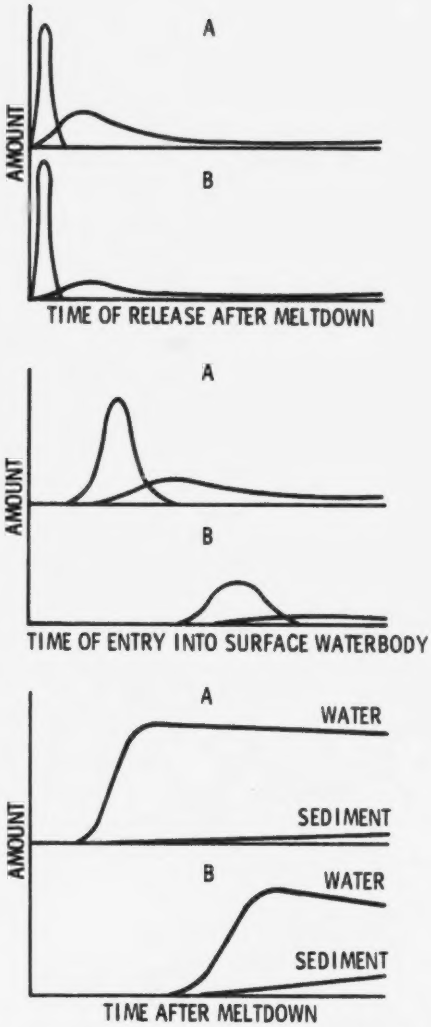

Figure 3.3 Schematic Depiction of the Behavior of Two Fictitious Nuclides, A and B, Which Differ Only in Their Leaching Rates and Distribution Coefficients; A Leaches More Rapidly and Has a Smaller Distribution Coefficient.

In addition to these movements on relatively massive scales, several phenomena would contribute to dispersal of materials entering surface waterbodies on a lesser scale. As in a subsurface waterbody, the net effect of all these phenomena would be to spread the radioactivity over a larger region than would otherwise be anticipated.

3.4.1 Lake

3.4.1.1 Water Movement

Naturally-occurring lakes range from small, rapidly flushing ones to large, slowly draining ones such as the Great Lakes. Approximately 15% of all the

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*Because the exposure of the human population at a given site would be roughly proportional to the amount of activity reaching the surface water, small differences in the groundwater travel times could likewise be associated with large differences in the anticipated population exposures.

U.S.'s nuclear reactors are located next to lakes. Each of the reactors located close to a lake is situated near one of the Great Lakes. Therefore, the Great Lakes are the lakes which are considered for this study.

In a Great Lake, water movement , and therefore transport of any containments carried by the water, occurs on several different scales. Usually there are currents close to and parallel to the shore which move one direction along the shore for several days. Then mixing with the offshore water results. This is followed by reversal of the alongshore current. The bulk of the surface water moves in lake-wide counterclockwise currents. Shoreline irregularities and uneven movements cause substantial dispersion on the background of all the various currents. Twice a year, in all except Lake Erie, seasonal turnover occurs in which the top 50 to 75 feet of water mixes with the rest of the lake. The net movement out of the lake by flow into the next lake results in flushing times for the Great Lakes varying from 3 years to 190 years (Table 3.1).

Table 3.1. Volumes and Residence Times for Some Waterbodies

System	Volume (m^3)	Residence Time (yr)
Lake Superior	1.2×10^{13}	189
Lake Michigan	4.9×10^{12}	31
Lake Erie	4.6×10^{11}	2.6
Lake Ontario	1.6×10^{12}	7.8
Reservoir	---	≤.01 to .1
Chesapeake Bay (estuary)	7.5×10^{10}	1
Hudson River Estuary	2.3×10^{9}	0.12
Eastern Continental Shelf	---	1.5
Tennessee-Ohio-Mississippi River System	---	0.38
Missouri-Mississippi River System	---	0.07

The Great Lakes are characterized by relatively rapid and substantial initial dilution of any material released into any one of them. The subsequent removal of the material by flow out of any lake is relatively slow. Consequently, concentrations of the dominant radionuclides from releases to the hydrosphere after a core-melt accident would be comparatively low in at least most of the lake but could persist for long times. Lower but still possibly significant concentrations might be found in succeeding lakes for even longer periods. Sizeable concentrations would be possible only in the vicinity of the accident.

3.4.1.2 Sediment Behavior and Sediment-Nuclide Interactions

Any of the Great Lakes can be regarded primarily as a sediment sink. The dominant source of the sediments is the erosion of the lands of the surrounding watersheds. The net sedimentation rate in any of the Great Lakes is typically rather low. Consequently, the removal of any radionuclides from solution by the sediments would be relatively slow. Permanent removal by burial would be expected only for the most strongly adsorbed radionuclides, because less strongly interacting radionuclides could reenter the water column by diffusion out of the sediments.

In general, the concentrations of the dominant radionuclides contributing to the population dose after a core-melt accident would be only slightly affected by sediment-nuclide interactions in a large lake such as a Great Lake. In any case, observable effects would usually occur only over extended periods of time.

3.4.2 Estuary

3.4.2.1 Water Movement

An estuary is essentially a semiclosed coastal body of water possessing a free connection with the open ocean and within which ocean water is noticeably diluted with fresh water.[1] Approximately 10% of the U.S.'s nuclear reactors are situated next to such waterbodies. An even larger percentage of all the reactors are located close to river systems (and other waterbodies) which eventually flow into significant estuaries.

Of the four basic types of estuaries,* a drowned-river valley estuary is thought of as being the classical type of estuary and is taken to be the representative type of estuary in this study. A drowned-river estuary is that portion of a river where the ocean tide meets the river current. As in any river, there is a net flow outward to the ocean. Because of tidal influences, the direction of the current reverses twice daily. The net seaward flow is usually relatively low. The effect of the tidal reversal is to cause significant dispersion in both upstream and downstream directions. In fact, in many estuaries the overall movement is characterized primarily by dispersion. The average residence time for water in an estuary can range from weeks to more than a year (Table 3.1).

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*Four primary subdivision of estuaries exist; drowned river valleys, fjord-type estuaries, bar-built estuaries and tectonically-produced estuaries.

The differences in density between river water and ocean water can result in stratification of water in an estuary due to salinity considerations.* Most estuaries have distinct partially-mixed horizontal layers. In such estuaries, the net flow in the bottom saline layer is toward the head of the estuary while the net flow in the surface freshwater layer is toward the estuary's mouth. There is a small net flow from the bottom layer to the surface one, with some interchange between the layers. A pollutant released into the bottom layer is first transported primarily upstream in the bottom layer, then downstream in the surface layer. In contrast, a pollutant released into the surface layer is primarily transported downstream in that layer; however, some of the material is dispersed into the lower layer and carried back upstream where it subsequently returns to the surface layer. As a result of the complex flow pattern in such estuaries, sizeable areas both above and below the site of the release could be contaminated.

After a meltdown accident near an estuary, the concentrations of the important radionuclides in the water would depend strongly upon both the turnover rate of water from the estuary and the rate(s) of the releases into that waterbody. For slow turnovers and rapid releases, those concentrations could be significant. Even after accidental releases into an upstream river, sizeable concentrations of some radionuclides could be found in certain estuaries.

3.4.2.2 Sediment Behavior and Sediment-Nuclide Interactions

Estuaries act as repositories for sediments. When rivers enter estuaries, much of the suspended sediment is deposited due to decreases in the water velocity. Inasmuch as rivers can carry large amounts of sediment, the sedimentation rate in an estuary can be quite high compared to that found in other types of waterbodies. The rate of sedimentation can be so high that even biological turnover and diffusion are inadequate to permit the return of at least some radionuclides to the water column.

The alternation of the direction of the water movement in estuaries can be accompanied by significant resuspension of sediment. Because resuspension can result in substantial transfer of activity from the water to the sediment, and vice versa, in very short periods of time, large quantities of radioactivity can be transported some distance relatively rapidly by such resuspension and subsequent movement of the suspended material.

As has been mentioned, the sediment-nuclide interactions in any waterbody are extremely dependent upon the physics and the chemistry of both the sediments and the water. As one proceeds toward the mouth of an estuary, the character of both the sediments and the water often changes drastically.

As a river proceeds into an estuary, the larger heavier sediment particles are deposited first; the smaller lighter sediment particles are deposited farther toward the mouth. In general, the smaller particles tend to have relatively more adsorptive sites and to be more efficient at removing radionuclides from the water column than larger particles are. Furthermore, they are much more likely to be resuspended. Consequently, the lower portion of an estuary is expected to be a sink for much of the radioactivity which enters the estuary.

Along with changes in sediment character which occur as a function of distance down the estuary, significant variations in the composition of the water also occur. The water in an estuary ranges from fresh at the upper end to saline at the lower end. As a result of these changes, chemical and physical reactions can take place during the passage down an estuary which can either substantially increase or decrease in concentration of certain elements in the water.*

In an estuary, the concentrations in the water of the dominant radionuclides released to the hydrosphere by a meltdown accident could be substantially altered by sediment-nuclide interactions due to both the relatively high sedimentation and resuspension rates and the significant residence time of the water. These interactions could be potentially more important in an estuary than in any other type of surface waterbody.

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*The amount and type of stratification depend both upon the relative magnitudes of the river flow and the tidal flow and upon the geometry of the estuary. Drowned-river estuaries range from the river-dominated, salt-wedge ones having distinct horizontal saline and nonsaline layers to tide-dominated, sectionally homogeneous ones having complete localized mixing. Intermediate between these two types are estuaries having distinct though partially-mixed horizontal layers and estuaries having distinct though partially-mixed vertical layers. As the magnitude of tidal flow increases relative to the river flow and/or as the width of the estuary increases relative to the depth, an estuary tends to become less stratified (more mixed).

*For example, certain species such as iron, aluminum and silicon, which tend to form gelatinous hydrates with increasing salinity, tend to precipitate out of solution with movement downstream. If the hydrate particles are too fine to be deposited directly out by sedimentation, they can be effectively removed from the water column by adsorption to either sediment particles or biological organisms. Suspended clay sediments can coagulate due to the saline conditions and be precipitated out of suspension; any adsorbed nuclides are precipitated with them. Alternatively, elements which tend to be highly adsorbed in fresh water can be almost completely desorbed in saline conditions.

3.4.3 Ocean

3.4.3.1 Water Movement

Wide ranges of descriptions are reasonable for coastal situations. Coastal sites vary from highly sheltered tidal bays of lagoons to fairly rapidly flushed, open coastal areas.* About 15% of the U.S.'s reactors are found close to all types of coastal sites. A relatively open region of the continental shelf can probably be taken to be a representative ocean setting provided that ocean sites with more restricted flushing are considered to be included with estuarial sites.

In an ocean, water movement in open coastal regions close to the shore resembles that in the nearshore waters in the Great Lakes. The primary differences in observed behavior of the two systems are the magnitudes and the time scales of the movements. Both are greater in the case of an ocean. In addition, tidal influences can be important in an ocean whereas they are not usually important in the Great Lakes.

If there is a release of radioactivity into the nearshore area of an open coastal region, local currents initially move the material away from the source. Typically, such nearshore currents periodically reverse direction, moving the radioactive material essentially back in the direction of its source. Over a long period of time, a low-level background of radioactivity is built up in the area of the source by the current reversals; superimposed on this background is any recent release. The water of the nearshore current gradually mixes with water of the other currents on the continental shelf so that the radioactivity becomes spread over the entire width of the shelf. The average residence time on the continental shelf generally ranges from months to a couple of years (Table 3.1)**

The concentrations of the radionuclides in the water after a meltdown accident would be relatively low on most of the continental shelf. Significant concentrations would be possible only in the vicinity of the accident and only if the release(s) occurred rapidly enough.

3.4.3.2 Sediment Behavior and Sediment-Nuclide Interactions

In most coastal regions other than estuarial ones, the net sedimentation rate is low compared to that found in most other waterbodies. Both the relatively rapid flushing and the low rate of sedimentation which occur in many open coastal regions combine to drastically limit the amount of sediment-nuclide interaction in such regions. Consequently, sediment interactions are not nearly as important at open sites which are not close to estuaries as they are at other types of sites.

In addition to the above-mentioned factors which tend to decrease the amount of sediment-water interactions, the high salinity of the ocean waters also tends to limit the amount of interaction. In general, the distribution coefficients of the radionuclides dominating the releases to the hydrosphere after a core-melt accident are much lower in salt water than they are in fresh water. Therefore, in salt water regions, especially in those with low sedimentation rates, the concentrations of the dominant radionuclides in the water would generally not be significantly lowered by sediment-nuclide interactions.

3.4.4 River System

3.4.4.1 Water Movement

Most nuclear reactors (60%) are located on river systems. Because of the large variation among the characteristics of those systems, it is extremely difficult to estimate what a representative system is. At least two distinct types of river systems, a free-flowing system and a heavily-dammed system, should probably be considered.

Water movement in a free-flowing river is characterized primarily by advective flow. However, many factors, such as shoreline irregularities and bends in the river, tend to cause uneveness in the flow. Consequently, noticeable dispersion can take place during flow down a free-flowing river.

In general, the rivers on which LWR's are situated are not free-flowing for their entire lengths. Instead, they have dams which tend to retard the flow of the water in certain stretches. The reservoirs behind the dams have flushing rates which are generally smaller than those of free-flowing portions of the river but are larger than those of comparably-sized natural lakes. Substantial mixing can occur within the reservoirs so that the dispersion of materials traveling down dammed river systems can be much greater than that for free-flowing systems. Flushing times for rivers vary from days to several months, with the shorter times being for free-flowing systems and the longer ones being for heavily-dammed systems (Table 3.1).

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*For the sheltered regions, tidal influences may dominate. The net velocity of the contaminated water can be very low in such situations, with tidal dispersion dominating the movement of the water. In such sheltered regions, all variations of the hydrological dispersion can exist, ranging from dispersion essentially only in one direction to equal dispersion in all directions. In open coastal regions, both the advection currents and dispersion can be important.

**The average residence time for water on the northeastern and northcentral portions of the American continental shelf is approximately 1-1/2 years. It is probably much less than this on the western shelf.

After a meltdown accident, the concentrations of the dominant radionuclides in the water would depend strongly upon both the rate(s) of the releases into the river and the rate of flushing of the river. The concentrations would also depend upon the size of the river. Significant concentrations could potentially be found in at least the upper reaches of many river systems after rapid enough releases to the hydrosphere.

3.4.4.2 Sediment Behavior and Sediment-Nuclide Interactions

Free-flowing sections of rivers tend to act as media for the transport of sediment to distant locations. In contrast to the free-flowing stretches, the reservoirs found in many rivers tend to act as sediment sinks.

Flow rates of rivers are extremely variable with time. As a result, the sedimentation rates also exhibit considerable variation. As in the case of an estuary, alterations in the velocity of the water tend to correspond to changes in the sizes and the types of sediment particles deposited, with the largest particles being dropped first. In addition, seasonal flooding can permit deposition of sediments in regions outside the normal river bed. Subsequent flooding can allow reincorporation of some of the previously deposited sediment back into the river channel proper.

The sediment-nuclide interactions in any given stretch of a river tend to be basically like those found in a lake of similar chemistry. In the river system taken as a whole, the interactions can vary over a wide range due to differences in the chemistries in various sections. In particular, the composition of both the sediment load and the river water can vary significantly due to differences in their sources.

In some river systems, the concentrations in the water of at least several of the radionuclides dominating the population dose after a meltdown accident could be significantly affected by sediment-nuclide interactions in relatively short periods of time. The effects would be most obvious in systems with reservoirs with both relatively long residence times and high sedimentation rates.

3.5 Dependencies

3.5.1 Dependencies in Aquifers

The consequences resulting from the transport of radioactivity through an aquifer are extremely dependent on both the flushing rate of the aquifer and strength of the soil-nuclide interactions. If exposure is through wells in the aquifer, then a slow flushing rate for the aquifer can increase the impact on the human population. However, if exposure is through surface water, then a slow flushing rate, and therefore a long groundwater travel time, for the aquifer can decrease the impact bv permitting substantial decay to occur before the surface water pathways are reached.

If the soil-nuclide interactions are strong,* then the travel of the nuclides through the aquifer will be relatively slow. In such a situation two primary factors can lower the impact on the human population: substantial decay can occur during the delayed transport; and the concentration in the water can be drastically lowered by "adsorption". The effects of prolonging a release into an aquifer are expected to be most noticeable when the prolongation permits decay to reduce the amount of radioactivity which reaches the population.

3.5.2 Dependencies in Surface Waterbodies

For any type of surface waterbody, the population exposure is rather dependent on the flushing rate of the waterbody. In contrast to the aquifer situation, the net exposure is only somewhat dependent on the strength of the sediment-nuclide interactions within the surface waterbody.

If the flushing rate is low, as for a Great Lake, then the amount of radioactivity in the water decreases relatively slowly with time. Consequently, the population exposure resulting from long-lived radionuclides can increase substantially with time. However, if the flushing rate is high, as for a free-flowing river, then the amount of radioactivity in the water decreases rapidly with time. As a result, the population exposure does not increase much with time.

If the sedimentation and resuspension rates are small, then the effects of sediment-nuclide interactions will generally be negligible. However, if those rates are very large, then the effects can be noticeable.

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*To be more exact, this should read "if the distribution coefficients are large".

3.5.3 Dependencies in General

The primary factors affecting the potential population dose resulting from releases to the hydrosphere after a core-melt accident are those determining the total amounts of the dominant radionuclides reaching the surface waterbody prior to decay.* In other words, they are the release rate(s) of the radioactivity into the groundwater and the travel times of the dominant radionuclides to the surface water.**,† The second most important factors affecting the population dose are the initial dilution rate for the affected surface waterbody and the appropriate flushing rate for that body. The factors usually of least importance but still of some significance are the sediment-nuclide interaction strengths and other sediment-related factors of the surface waterbody.

Any factor which delays contact of the radioactivity with the human population not only decreases the amount of radioactivity available (by decay) but also increases the chance that procedures can be instituted to prevent the contact of radioactivity by the population. Such interdictive procedures can either completely preclude or else somewhat lessen the human exposure to the radioactivity by removing all or some of the radioactivity from effective interaction with the hydrosphere. The types of interdictive procedures which can be employed are discussed in Appendix F. The effects of taking such mitigating actions are considered in Chapters 6 and 7.

3.6 Summary

For any liquid pathway, the most important consideration in estimating the risk to the human population is the time that it takes for a radionuclide to reach a point at which exposure can take place. To reach such a point, any radionuclide must travel from the location of the core-melt accident to either the closest surface waterbody or the closest well. As the radionuclides travel with the groundwater, they undergo radioactive decay. As a result, the longer the underground travel time to the point to potential exposure is, the less risk there is to the human population.

The underground travel time for a nuclide is determined primarily by the combination of two factors: the time it takes for groundwater to move from the location of the accident to the nearest significant surface waterbody or well; and the rate at which the nuclide moves relative to the speed of the groundwater. The rate at which the groundwater moves depends upon the hydraulic gradient of the site and upon both the permeability and the effective porosity of the substrate. The estimated groundwater velocities beneath the reactor sites vary from 10^{-4} to 10^{3} m/da. The associated groundwater travel times to the nearest surface waterbodies vary from less than 1 day to more than 10^{6} days.

The relative rate at which any radionuclide moves through the ground depends upon the chemical and physical properties of both the nuclide and the substrate. The movement of the nuclides can be significantly delayed by interactions with the soil substrate. These interactions can result in the emergence of some radionuclides at times up to several orders of magnitude later than the times at which the first nuclides emerge. For long enough net travel times, only the very long-lived radionuclides can have any effect on the human population. Any factors which tended to delay contact of Sr and Cs with accessible portions of the environment for more than 100 years would substantially reduce the possible population dose resulting from any releases to the hydrosphere after a meltdown accident.

Once introduced into a surface waterbody, radionuclides can attach themselves to the sediment, they can enter the foodweb, they can remain dissolved or suspended in the water, or they can be continuously recycled through some subset of these. Eventually any radionuclide will be removed from further effective interaction with pathways to the human population. The period over which a radionuclide can be important in a specific waterbody is given essentially by its environmental residence time. These times range from approximately 10^{-3} years for some radionuclides in certain rivers to 10^{6} years for some other radionuclides in noncoastal portions of the oceans. Some relevant times are given in Table 3.2.

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*The details of all the dependencies are presented in Section B4 of Appendix B. The interested reader should review the summary of that discussion presented in Subsection B4.5.

**The possible values of the release periods and the groundwater travel times, and hence any associated delay times, vary over orders of magnitude. Thus, the amount of decay of the important radionuclides which occurs prior to contact with the human population covers a broad range. As a result, the related potential consequences vary by substantial amounts.

†In general, the effects of Sr-90 and Cs-137 would tend to dominate the consequences to the human population resulting from most releases to the hydrosphere after a meltdown accident. (The extent of this dominance is substantial for most of the time within 100 years after the accident.) Therefore, any factor or combination of factors which tended to delay contact of these elements with any wells or surface waterbodies beyond approximately 100 years would tend to significantly reduce the resultant radiation dose to the population.

Table 3.2 Comparative Radionuclide Transport Properties of Waterbodies*

Type of Waterbody	Dissolved Phase Transport	Sediment Effects	Residence Time of Water	Residence Time of Nuclides (for instantaneous release)
Aquifer	Poor dilution; radioactivity confined to relatively small region.	Soil acts as an effective temporary sink for many radionuclides, because is immobile, it can greatly delay the emergence of these sorbed nuclides into nearby surface waterbodies.	Days to millions of years, depending upon the hydrogeological conditions.	Days to millions of years or even longer, depending upon the hydrogeological conditions and upon the specific nuclides.
Great Lake	Rapid initial transport by large-scale turbulence; seasonal mixing by turnover of thermally stratified layers; flushing out of lake slow because of large volume and relatively small flow rate.	Nearly all sediment entering lake is trapped; very effective sink for some radioactivity; sediment resuspension is minor - largely confined to coastal areas during storms.	Years to tens of years.	Years to tens of years for dissolved phase once completely mixed in lake; virtually forever (decay limited) for sediment phase.
Estuary	Net removal from system usually much slower than in rivers; stratification can occur due to variations in salinity; dispersion from tidal movements is significant; saltwater-freshwater interface causes deposition from solution of many radionuclides.	Lower parts of most estuaries are sediment traps, with virtually all sediment being deposited; upper portions may have net downstream sediment transport; major resuspension of sediment possible during storms; some resuspension by tidal oscillations; rapid sediment deposition can remove contaminated sediments from further contact with water.	Days to months.	Weeks to years in dissolved phase: indefinite in sediment phase.
Ocean	Rapid dispersion in large-scale coastal turbulence; unconstrained by boundaries other than shoreline; both thermal and saline stratification occurs; some vertical seasonal mixing occurs.	Sediment interaction minor because of low rate of sedimentation off shore.	Hundreds of days on continental shelf; millions of years in open ocean.	Hundreds of days on continental shelf; millions of years in open ocean.
Small Lake	Rapid initial transport within lake; seasonal mixing by turnover of thermally stratified layers; flushing is usually faster than large lake due to the smaller volume.	Nearly all sediment entering lake is trapped; very effective sink for some radioactivity; little sediment resuspension.	Months to years.	Months to years for dissolved phase; virtually forever (decay limited) for sediment phase.
Reservoir on River	Relatively rapid initial transport within reservoir; seasonal mixing by turnover of thermally stratified layers; flushing is generally much more rapid than for a comparably-sized natural lake; dispersion from passage through a reservoir can be substantial.	Much sediment trapped, except during floods or dredging; sediment is an important sink for some radioactivity; sediments may also be occasionallly resuspended; rapid sediment deposition can remove contaminated sediments from further contact with water.	Weeks to months.	Weeks to months in dissolved phase; sediment phase indefinite, probably longer than for fast flowing river.
Free-flowing River	Moves at relatively high velocity; some dispersion.	Fine sediments, which contain most of sorbed radioactivity, will be suspended during high flow periods; low flows will allow settling; contaminated sediments may be frequently resuspended by high flows.	Hours to days.	Hours to days in dissolved phase; indefinite in sediment phase.

*Adapted from Table 4.2.5 in NUREG-440 "Liquid Pathway Generic Study," NRC, 1978.

The residence time of a radionuclide represents the combined effects of all the possible fates of that radionuclide. Those fates include the ones mentioned above plus removal by flushing. Flushing is important for all waterbodies except noncoastal regions of the oceans; oceans can reasonably be regarded as the ultimate sinks for the radionuclides. If no significant interaction occurs with any portion of the environment, then the residence time of a radionuclide is essentially equal to that of the water. Most interactions tend to prolong the effective residence time relative to that of water.

For very few, if any, reactor sites would significant short-term human exposure be expected to occur as a result of direct use of contaminated aquifer water. The most likely pathway to humans would involve exposure from either direct or indirect contact with polluted surface water. For most sites, it is expected to be many years before significant quantities of radionuclides could reach accessible portions of the environment via groundwater transport.

Substantial site-to-site variation of the consequences can be expected due to the large differences in groundwater travel times. Some additional site-to-site variation can be expected as a result of the differences in both the relative travel rates of the dominant radionuclides (in the ground) and the relevant surface waterbody characteristics (especially the flushing and dilution rates).

References

[1] D. W. Pritchard, *"What is an Estuary: A Physical Viewpoint,"* in Estuaries, Publication No. 83, American Association for the Advancement of Science, 3, 1967.

4 Pathways to the Human Population

4.1 Introduction

The radiation doses received as a result of the release of radioactivity to the hydrosphere following a core-melt accident would depend upon the concentrations of the radionuclides in the relevant liquid exposure pathways. These concentrations would, in turn, depend on the corresponding concentrations in the water.

Therefore, the behavior of the radionuclides in each of the liquid exposure pathways would be a function of both the type of waterbody(ies) under consideration and certain pathway characteristics. The dependence upon the type of waterbody would arise from the transport of the radionuclides within the body; the relevant characteristics of the waterbody would be its dilution and flushing properties, along with the behavior and interactions of its sediment. The dependence upon the pathway characteristics would be due to the complicated movement of the radionuclides within some of the pathways; the relevant factors would be the time-dependent uptake and removal of the radionuclides by some portions of the environment, along with the interconnectedness of various parts of that environment. The dependence upon transport in the hydrosphere was considered in the previous chapter; the dependence upon movement within the pathways is reviewed here.

In this chapter, the dominant liquid pathways by which radioactivity might reach the human population are considered. In Section 4.2, the general effects of the complexities of the interactions of the radionuclides within some of the major liquid exposure pathways are described. In Section 4.3, the dominant pathways to the human population are considered as functions of the hydrological site types. In Section 4.4, the most important parameters for the pathways involving extensive time-dependent interactions within the environment are reviewed. In Section 4.5, the relevant time scales for each of the pathways are covered. The last section of this chapter, Section 4.6, contains a comparison of the relative importance of each of the pathways as a function of hydrological site type.

Qualitative discussions of the factors affecting the resultant radiation doses received from each of the liquid exposure pathways are presented in Appendix E; in particular, those factors which relate to the movement of the radioactivity within each of the pathways are considered. The related pathway models used in this report are described in detail in Appendices C and D. The estimated popuiations-at-risk are considered in Appendix E.

4.2 Pathways

4.2.1 Important Factors

Unlike the dominant atmospheric exposure pathways, at least three potentially significant hydrospheric exposure pathways would involve movement of the radioactivity in the environment via relatively complicated sets of interactions. As a result, the concentrations of the radionuclides in those liquid pathways would not necessarily be straightforwardly related to the concentrations in the hydrosphere.

Unfortunately, the adequate consideration of the movement of radioactivity within the exposure pathways is generally much more difficult than that of transport in the hydrosphere itself. In transport through any surface waterbody, the pathway interactions can usually be regarded as being only negligibly perturbing effects with respect to the concentrations in the water. However, it is those same very irregular and extremely complicated interactions which are the dominant determinants of the concentrations in the relevant portions of the pathways.

4.2.2 Pathway Concentrations

In this study, the five liquid exposure pathways considered are the following: ingestion of drinking water; ingestion of aquatic foods; ingestion of irrigated foods and related animal products and by-products; shoreline usage; and immersion in water. Three of those pathways involve extensive time-dependent interactions of the nuclides with parts of the environment other than just the water. They are the two including ingestion of food and the one considering shoreline usage.

In all three, the radioactivity builds up gradually in the exposure pathways after entry of the radionuclides into the water. Likewise, after the source of contamination is terminated, radioactivity frequently persists in these pathways for some time. The buildup in and the removal from these pathways are not necessarily smooth functions with respect to either time or the concentrations of the radionuclides in the water.

For steady releases of radioactivity into the environment, the amounts of certain radionuclides in the

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three time-dependent pathways sometimes tend to build up to "equilibrium" concentrations. These concentrations of radioactivity in the three pathways are often much large than the corresponding concentrations in the water. For the two food pathways, the equilibrium concentration for a given radionuclide is given approximately by the product of the concentration in the water and a factor called the concentration factor; for the shoreline pathway, the equilibrium concentration is given approximately by the product of the concentration in the water and the distribution coefficient of the shoreline sediment.

The time-dependent behavior in the three pathways is generally most noticeable for situations in which there are both relatively rapid changes in the concentrations of activity in the water and comparatively complicated networks of interactions by which that activity moves through the relevant portions of the environment. For a rapid buildup in the water, the pathway concentration initially may be far from "equilibrium." It may take quite some time for the pathway concentration to approach the equilibrium concentration (Figure 4.1a). In contrast, for a comparatively slow buildup of activity in the water, the pathway concentration is often not far from its equilibrium value at any time even for a relatively complex pathway (Figure 4.1b).

There would be at least two important effects on the estimated consequences of a meltdown accident resulting from the complexities of the movement of radionuclides in accessible portions of the environment. The primary result of all the time delays caused by the interactions within some of the liquid exposure pathways would be to cause a shift to later times of the period over which radiation exposure would occur relative to that expected by consideration of the concentrations of radioactivity in just the water. For releases to the hydrosphere of relatively long duration, these shifts would probably not be significant on an overall population dose basis. However, they might be noticeable if the temporal dependence of that population dose were important. In contrast, for relatively rapid releases, the shifts might be noticeable for determining both the net population dose and the rate at which that dose were delivered.

The other potentially important effect resulting from the complexity of the interactions could be the displacement of the pathway concentrations from those expected by simpler considerations. Such displacement could occur even for relatively slow steady releases.

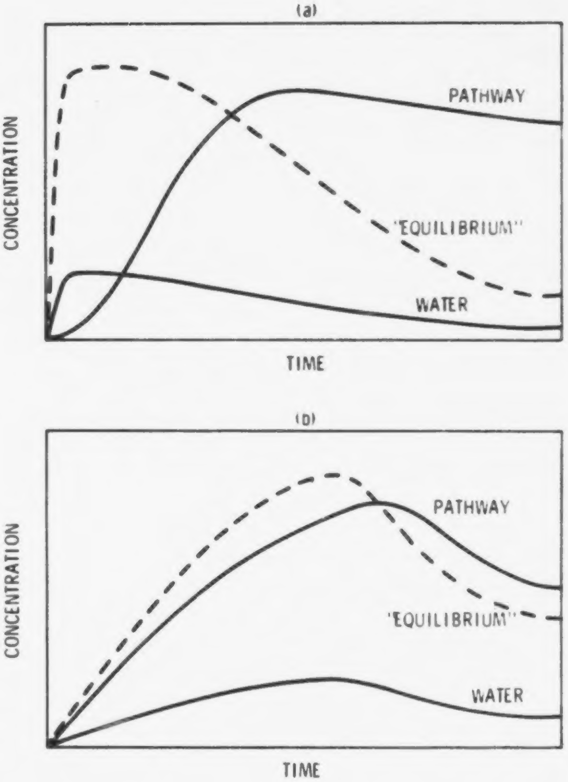

Figure 4.1 Schematic Diagram of the Concentrations of Radioactivity in Two Pathways: One with Time-Dependent Buildup and Removal and Another with Concentration-Factor Behavior. (Part (a) represents a situation for a relatively rapid change of concentration of radioactivity in the water and Part (b) illustrates a situation for slow change of concentration.)

4.3 Pathways as Functions of Site Types

The detailed descriptions of the movement of the radioactivity within the various exposure pathways is extremely dependent upon the characteristics of the nearest waterbody. In a zeroth-order approximation, that movement depends primarily upon the basic classification of the waterbody as either a large lake, an estuary, an ocean or a river system.

The locations of all constructed, all under-construction and some proposed US nuclear reactors are depicted in Figure 4.2[1] and are listed in Table B1 of Appendix B. For the sites shown in Figure 4.2, 22 reactors are on Great Lakes, 17 are on estuaries, 24 are on ocean coasts, and 122 are on river systems. Of

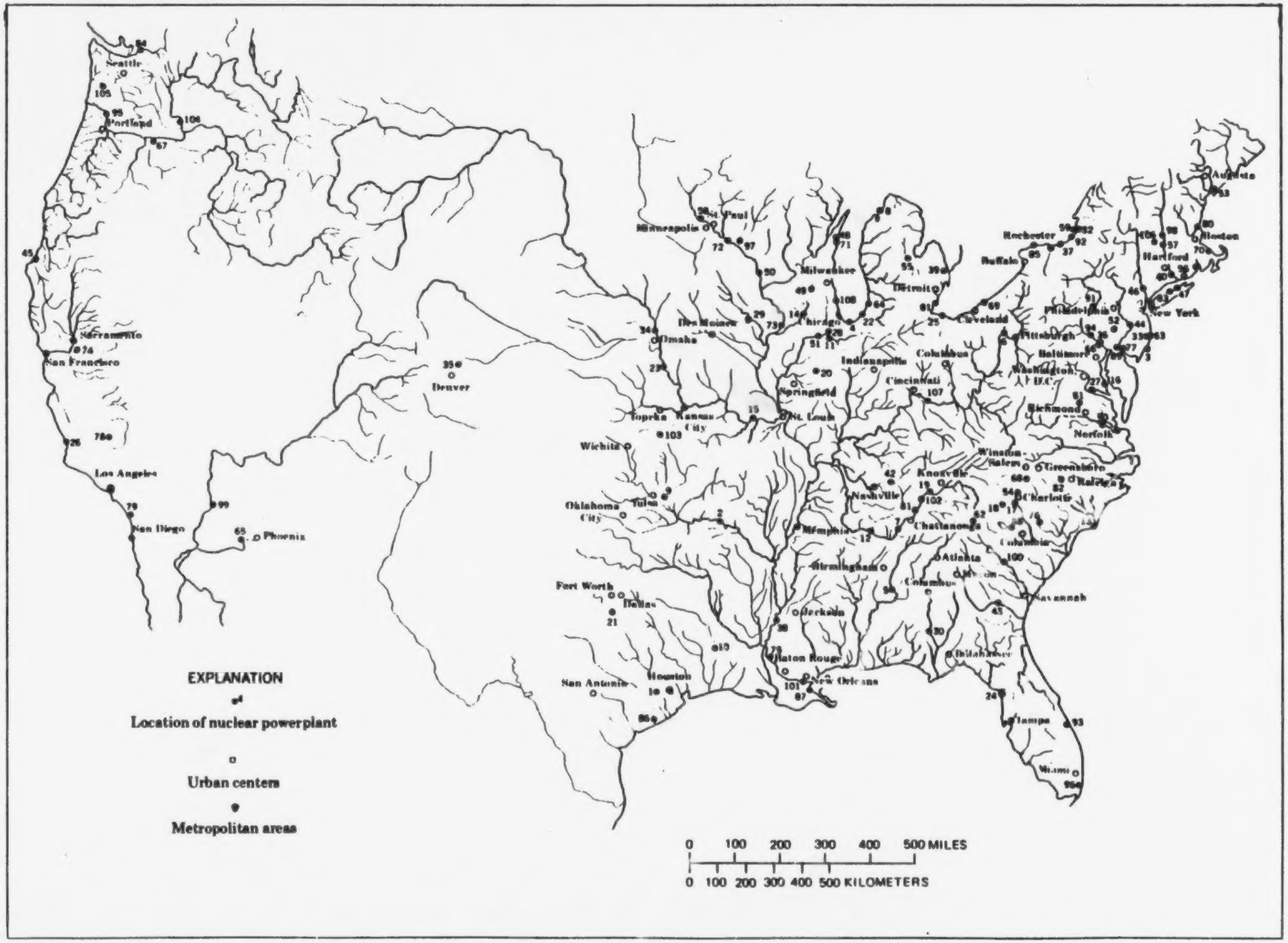
Seattle
Portland
Sacramento
San Francisco
Los Angeles
San Diego
Phoenix
Denver
Minneapolis
St. Paul
Des Moines
Omaha
Topeka
Kansas City
Wichita
Oklahoma City
Tulsa
Fort Worth
Dallas
San Antonio
Houston
Milwaukee
Chicago
Detroit
Springfield
St. Louis
Indianapolis
Cincinnati
Columbus
Cleveland
Buffalo
Rochester
Pittsburgh
Philadelphia
New York
Hartford
Boston
Augusta
Baltimore
Washington, D.C.
Richmond
Norfolk
Winston-Salem
Greensboro
Raleigh
Charlotte
Knoxville
Nashville
Memphis
Chattanooga
Birmingham
Atlanta
Columbus
Columbia
Savannah
Jackson
Baton Rouge
New Orleans
Tallahassee
Tampa
Miami
EXPLANATION
Location of nuclear powerplant
Urban centers
Metropolitan areas
0 100 200 300 400 500 MILES
0 100 200 300 400 500 KILOMETERS

Figure 4.2 Location of U.S. Nuclear Power Plants, Including Existing, Under-Construction and Proposed Plants.

these, 13 on Great Lake sites, 8 on estuary sites, 10 on ocean sites, and 57 on river system sites can probably be classified as "dry" sites.* (Of the more restricted list of 100 reactors considered in the Reactor Safety Study, 16 are on Great Lakes, 7 are on estuaries, 15 are on oceans, 45 are on river systems, and 17 are at dry sites (see Table E1 in Appendix E).)

Some of the more important environmental and other factors affecting the radiation doses which would be received by individuals and by the entire population at each type of waterbody after a meltdown accident are summarized in Table 4.1 for each of the five pathways considered.**,† These factors are discussed in more detail in the remainder of this section.††

4.3.1 Drinking Water

The magnitude of the total drinking water ingestion exposure due to an accident at a given type of hydrological site would depend upon several factors, including the type of water at the site (salt or fresh), the rates of dilution and flushing of the contaminated waterbody, the strengths of the sediment-nuclide interactions, the size of the affected drinking water population, and the type(s) of water processing employed. There would be no drinking water pathway at all ocean sites and at many estuary sites. At all freshwater sites, the drinking water pathway usually would be one of the more important means of human exposure after releases of radioactivity to the hydrosphere.

4.3.1.1 Dry Sites

Because the local concentrations of any radioactivity would be expected to be orders of magnitude higher in a contaminated aquifer than they would be in any surface waterbodies, the resulting doses to individuals could be very large. However, the size of the total drinking water population at an aquifer would be expected to be quite small. Only those sites classified as "dry" would have a potential drinking water pathway via aquifer usage. Yet, not even many of those sites would actually have such a drinking water pathway (see Appendix E).

In general, reactors tend to be located in what could be classified as rural settings. Typically, the water taken from aquifers in rural areas undergoes only minimal treatment. Consequently, at the few sites which did have a drinking water pathway, little radioactivity would normally be removed by processing.

4.3.1.2 Great Lakes Sites

Although the concentrations of activity in any of the Great Lakes would be rather low, the drinking water population could be so large that a significant net population exposure could occur. Furthermore, the very long residence time of some of the radioactivity in the water could contribute to increasing the potential total population dose. In contrast to the dose received by the total population, the doses received by individuals would be comparatively small except perhaps in the region close to the accident.

Most of the water taken from the Great Lakes probably undergoes some type of treatment. This is especially the case for water to be used for large population centers. Because the suspended sediment burden of the Great Lakes water is generally low, the amount of radioactivity removed from Great Lakes water by filtering alone would usually not be significant. However, noticeable amounts of radionuclides in solution could be removed by the more extensive processing which is often employed.

4.3.1.3 Estuarine Sites

Typically, the water in estuaries is too saline for direct consumption. However, drinking water can be taken from the upper portions of salt-wedge estuaries. The complex hydrological transport which occurs in estuaries could result in some contamination of the fresh water, even if the release of radionuclides were into the downstream saline portion of the estuary. However, the total drinking water exposure which could result from such releases into saline portions is thought to be small.

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*This classification ignores the effects of construction at the sites. If such effects were taken into account, the number of potentially dry sites would probably be much smaller.

**It should be noted that in addition to the radiation dose received via the initially impacted waterbody, additional exposure could occur at any waterbody downstream from the contaminated waterbody. These effects are not included in the following discussions of the individual waterbodies but should be kept in mind. The additional radiation dose received from secondary contamination of other waterbodies would generally be most significant for flow into either Great Lakes or estuaries.

†Radioactivity residing in any portion of the hydrosphere generally tends to be diluted whereas radioactivity residing in any of the three more complicated exposure pathways tends to be concentrated. As a result, even though the concentrations of radioactivity in the water may be low, the corresponding concentrations in the contacted portions of the exposure pathways may be significant.

††Closely related discussions emphasizing the procedures for mitigating the doses potentially received from each of the pathways are presented in Subsection F3.2 of Appendix F.

Table 4.1 Factors Affecting Potential Impacts of Various Exposure Pathways

Site	Large Lake	Estuary	Coastal Shelf	River System	Dry Site
Relative Concentration in Water	Low	Moderate	Low	Low to moderate	High
Pathway					
Drinking Water	Substantial treatment typical especially for highly populated regions	Little drinking usage	—	Substantial treatment typical, especially for turbid stretches	Treatment often minimal, especially in rural areas
Aquatic Food	Shellfish harvest minimal; finfish harvest substantial	Both shellfish and finfish harvests potentially large; large number of migratory finfish	Both shellfish and finfish harvests potentially large; large number of migratory finfish	Only finfish harvest usually signfiicant	—
Irrigation	Total amount of water for irrigation small	Little irrigation water usage	—	Amount of water for irrigation sometimes substantial	Amount of water for irrigation from aquifers sometimes substantial-but not at most, if not at all, dry sites
Shoreline Usage	Some flooding and wave action on narrow beaches; shoreline sediments not overly "sticky"	Both flooding and tidal action; shoreline and floodplain sediments relatively adsorbant	Both flooding and wave action on beaches; shoreline sediments not too "sticky"	Flooding of broad floodplains; shoreline and floodplain sediments relatively	—

4.3.1.4 River System Sites

Although the total time of exposure after a release of short duration might be comparatively small, the net population exposure could be large due to both the long lengths of some river systems and the large drinking water populations found along sections of certain systems. Furthermore, the concentrations of activity could be significant in at least the upper portions of some river systems if the rate of release into the hydrosphere were large enough. Consequently, both the total population dose and the related individual doses for drinking water ingestion could be substantial.

For those rivers which are relatively turbid, the drinking water is frequently processed rather thoroughly. Consequently, most of the radioactivity residing on suspended sediment and at least some of the radioactivity in solution would normally be removed by processing. For less turbid water, the drinking water is often not processed as completely. As a result, proportionately less radioactivity might be removed by processing such water than might be removed by treating more turbid water.

4.3.2 Aquatic Foods

The magnitude of the total exposure resulting from consumption of aquatic foods after an accident at a given type of hydrological site would depend upon several factors, including the type of water (salt or fresh), the rates of flushing and dilution of the contaminated waterbody, the strengths and the magnitudes of sediment-nuclide interactions, the types of aquatic organisms growing in the vicinity, and the sizes of the aquatic harvests. There would be no aquatic food pathway at a "dry" site. At all sites, except the "dry" sites, the aquatic food pathway would be one of the more important means of human exposure after releases of radioactivity to the hydrosphere; at most saltwater sites, it would probably be the most important pathway. (More detailed discussions of the aquatic food pathways at the various types of surface-water sites than those which follow are given in Section C3 in Appendix C.)

4.3.2.1 Great Lakes Sites

After releases of either short or long duration, the slow flushing of these lakes would tend to permit the accumulation of "maximum" concentrations of most radionuclides in the majority of the aquatic organisms harvested from the Great Lakes. However, the magnitudes of even those "maximum" concentrations would tend to be relatively small due to the correspondingly low concentrations in the water.

The harvests from the Great Lakes consist almost entirely of finfish. In fresh water, such as that in the Great Lakes, finfish tend to substantially accumulate all of the radionuclides which would dominate the releases to the hydrosphere after a core-melt accident. Because these harvests are large, the total population dose due to aquatic food ingestion could be significant for an accident at a Great Lakes site. In contrast, the related individual doses would generally be relatively small. However, a few persons ingesting finfish residing primarily close to the accident site could perhaps receive significant individual doses.

4.3.2.2 Estuarine Sites

The flushing from an estuary would often occur slowly enough to permit substantial buildup of at least certain radionuclides in many, if not most, estuarine organisms after radioactive releases of any duration. If the release of activity into an estuary were rapid enough, then the resultant concentrations in the resident aquatic organisms could be relatively large.

Estuaries tend to have relatively abundant animal, and sometimes plant, life. Besides the resident populations, there frequently exist rather large populations of transient organisms which pass through the estuary either regularly to feed or occasionally to spawn. The concentrations of radioactivity in many of these transients would probably be much less than those in comparable permanent estuarine residents; however, the transients could still contribute a substantial fraction of the total aquatic food dose to humans due to the potentially enormous number of such transients.

In salt water, such as that present in at least the lower portions of estuaries, aquatic organisms tend to accumulate only some of the dominant radionuclides which would be released to the hydrosphere by a core-melt accident. However, the total harvests from estuaries are sometimes enormous and substantial fractions of some of these catches are composed of members of lower phyla, such as crustaceans and mollusks.* In addition, many of the dominant radionuclides would have relatively long residence times in the aquatic foodweb due to their persistence in the sediments. Thus the potential exists that significant population and individual exposures could both occur via consumption of aquatic food from contaminated estuaries.

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*Many radionuclides tend to be much more concentrated in less highly evolved species.

4.3.2.3 Ocean Sites

Any radioactive contamination would be dispersed over a wide area and could contaminate very large quantities of seafood to low levels. After a release of short duration, the rapid flushing would prevent the maximum (equilibrium) concentrations of activity from accumulating in many aquatic species consumed by humans. However, a steady release of relatively long duration might permit such concentrations of activity to build up in many of these species. Even these concentrations would usually be relatively small because of correspondingly low concentrations in the water.

Ocean harvests consist of finfish, as well as significant quantities of crustaceans, mollusks, and, in some areas, kelp. As in esturaries, both resident species and migratory species of finfish might be contaminated. Appropriate consideration of the transients could be very important in determining the total dose received by the human population because a substantial fraction of the commercial marine finfish harvest in some regions is from migratory species.

As was mentioned in the discussion of estuaries, the dominant radionuclides tend to be less concentrated in marine organisms than in freshwater ones. However, because the aquatic food harvest is so large in certain areas and because a significant fraction of the harvest often consists of members of lower phyla, a substantial population exposure might be received via aquatic food consumption after a prolonged release. In contrast, the corresponding individual exposures from aquatic foods would generally be relatively small. Only persons ingesting aquatic organisms which resided near the site of the accident could receive significant individual doses.

4.3.2.4 River System Sites

After a release of short duration into a free-flowing river system, rapid flushing would probably preclude the higher members of the relevant foodchains from attaining "equilibrium" concentrations of many radionuclides. However, after either releases of somewhat longer duration into such systems or any releases into heavily dammed river systems, the higher members could attain such concentrations which could be significant.

Some rivers are important sources of finfish for human consumption, although many rivers produce only small catches of finfish for human use. Additional exposure could result from contamination of finfish species migrating into the river to spawn. In general, only negligible amounts of shellfish are harvested from river systems. For rivers with large harvests of aquatic foods, the total population dose due to aquatic food ingestion could be significant after somewhat prolonged releases. Furthermore, the corresponding doses to some individuals from this pathway could also be substantial.

4.3.3 Irrigated Foods

The magnitude of the total exposure resulting from consumption of irrigated foods and related animal products after an accident at a given hydrological site would depend upon several factors, including the rates of dilution and flushing of the contaminated waterbody, the volumes of water utilized for irrigation, the types of crops grown by irrigation, the sizes of the harvests of irrigated foods and the ultimate dispositions of the irrigated crops. There would be no irrigation pathway at saltwater sites. In general, of the pathways considered, the irrigation pathway would be one of the less important means of human exposure after releases of radioactivity to the hydrosphere. It would usually be significant only at certain dry and river sites. Even at those sites, only the total population dose could be substantial; the corresponding individual doses would be small.

4.3.3.1 Dry Sites

Substantial contamination of crops could occur through irrigation at a given "dry" site. However, the total number of crops irrigated using water from the contaminated part of an aquifer is expected to be relatively small. And even more important, most "dry" sites would not experience any usage of contaminated water for irrigation. Although irrigation might be an important pathway at a few specific "dry" sites, it is not expected to be significant in general.

4.3.3.2 Great Lakes Sites

The potential exists for irrigating large numbers of crops with slightly contaminated water. However, the region in which the Great Lakes are situated typically receives adequate rainfall for most of the crops grown in the vicinity. As a result, only relatively small amounts of water are used for irrigation in the Great Lakes region. Therefore, in general, the irrigation pathway is not expected to be significant for Great Lakes sites.

4.3.3.3 Estuarine Sites

Use of water for irrigation from estuaries is generally negligible. Consequently, the irrigation pathway is not expected to be significant for estuarine sites.

4.3.3.4 River System Sites

The potential exists for irrigating large numbers of crops with moderately contaminated water. Many of the river sites are located on river systems from which little water is taken for irrigation. However, a sizeable group of existing and proposed sites are located on systems from which substantial amounts of water are used for irrigation. Consequently, one might expect that reactors located on such river systems would have the most significant irrigation pathways (see Section E5 in Appendix E).

4.3.4 Shoreline Usage

The magnitude of the total shoreline exposure after an accident at a given type of hydrological site would depend upon several factors, including the rates of flushing and dilution of the contaminated waterbody, the types of sediment particles deposited upon the shores, the types of deposition and weathering mechanisms which were important, and the size of the affected population. There would be no shoreline exposure at a dry site, unless one considered the exposure due to contaminated irrigated fields. Of the pathways considered, the shoreline pathway would generally be one of the less important means of human exposure after releases of radioactivity to the hydrosphere.* However, it would always be the most important external exposure pathway.

4.3.4.1 Great Lakes Sites

In general, the concentrations of radionuclides in the sediment particles along the Great Lakes' shores would be relatively low. Furthermore, the width of the affected shoreline would generally not be too large. However, the exposed population at a Great Lakes site could be rather large. The relatively low concentration of radioactivity in the sediments would probably result in a comparatively low shoreline population exposure from a release at a Great Lakes site. Likewise, the corresponding individual exposures would be relatively low, except perhaps in the immediate vicinity of the release.

4.3.4.2 Estuarine Sites

The concentrations of radionuclides in the sediment along the shore of an estuary could be relatively high. This is partly because of the adsorptive properties of some of the sediment and partly because of the flooding and the tidal reversals which occur. Furthermore, the exposed population could be comparatively large at some estuaries. Consequently, relatively large shoreline population and individual exposures could both occur along an estuary.

4.3.4.3 Ocean Sites

The concentrations of radionuclides in the sediment particles along the ocean coasts would generally be relatively low. However, the exposed population could be rather large. The very low concentration of activity in the sediment would generally result in a very low population shoreline exposure at an ocean site. Similarly, the related individual exposures would be relatively low, except maybe in the immediate proximity of the accident site.

4.3.4.4 River System Sites

The concentrations of radionuclides in the sediment particles along the shores and on the floodplains could be relatively high. However, flooding in a river system could spread the contaminated sediment over a very wide area. Because of the potentially substantial amount of radioactive material in the sediment and because of the large population which could be exposed, a considerable shoreline population exposure could occur along a river system. Individual exposures would depend upon the concentration of the radioactivity in the sediment and not just on the total amount of activity. In some cases, the individual exposures could be significant.

4.3.5 Immersion

The magnitude of the total immersion exposure after an accident at a given type of hydrological site would depend, at least, upon the flushing properties of the surface waterbody and the size of the affected population. There would be no immersion pathway at dry sites. Of the pathways considered, the immersion pathway would generally be the least important means of human exposure after releases of radioactivity to the hydrosphere. Both the total population dose and the related individual doses for this pathway would be relatively small at any site.

4.3.5.1 Great Lakes Sites

The concentrations of radionuclides in the Great Lakes would be very low. However, the exposed populations for most immersion-related activities could be quite large. As a result, the population immersion dose for the Great Lakes site could be large relative to the immersion dose at other types of sites.

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*The model used to estimate shoreline exposure appears to have little correspondence to reality, particularly in a generic sense. The estimates of the shoreline doses given by this model are significant (see Chapter 6) but are thought to be relatively conservative, especially with respect to estimating individual doses.

4.3.5.2 Estuarine Sites

The concentrations of radionuclides could be relatively high for estuaries with low flushing rates. However, estuaries tend to be rather turbid. Therefore, they tend to have little swimming along their lengths. There can be substantial amounts of other immersion-related activities, such as boating, fishing and waterfowl hunting. A relatively small population immersion exposure would probably result along an estuary.

4.3.5.3 Ocean Sites

The total population receiving immersion exposure at oceans could be large. Nonetheless, the low concentrations and the low residence time of the activity in regions of potential exposure would probably combine to result in a relatively small total immersion exposure at a typical ocean site.

4.3.5.4 River System Sites

The local concentrations of radionuclides in the water in river systems could be relatively high. However, as for an estuary, the swimming exposure is expected to be relatively low. Therefore, a relatively small total immersion exposure would result along most river systems.

4.4 Pathway Dependencies

The radiation dose resulting from a given pathway depends upon the concentration of radioactivity in the pathway. In general, that concentration can depend on both the concentration of activity in the water and the history of that concentration in the waterbody, along with the time-dependent behavior of the relevant portions of the environment. The activity in two pathways, drinking water and immersion, depends essentially only on the concentration in the water. However, as has already been mentioned, the concentration in the other three pathways depends strongly on several factors.

The most important time-dependent factors in determining the concentrations of activity in the three pathways which include extensive time-dependent movement of the radionuclides in portions of the environment include the following: the uptake and elimination rates of radioactivity by aquatic organisms, irrigated plants, and animals fed with irrigated plants; the growing conditions for all three types of organisms; the interaction rates of radionuclides with sediment particles; and the deposition and weathering rates for shoreline sediment. The most important time-independent factors for these pathways include the concentration factors for aquatic organisms, irrigated plants, and farm animals, and the distribution coefficients for the shoreline sediments.*

4.5 Time Scales

The relevant time scales for the movement of the radionuclides along each of the pathways include the following: the times required for the radionuclides to build up in the pathway to significant levels; the times the radionuclides remain in the pathway after the radionuclides have been diluted or flushed from the water; and the delay times in the pathway due to processing procedures. Estimates for these times are presented in Table 4.2.

4.6 Summary

There are five potentially major pathways to the human population: ingestion of drinking water; ingestion of aquatic foods; ingestion of irrigated foods and related animal products and by-products; external shoreline exposure; and external immersion exposure. Three of these pathways involve complicated time-dependent movement of the radionuclides with certain portions of the environment.

The concentrations of radioactivity in the portions of the three complicated pathways which are contacted by humans are often much larger than the corresponding concentrations in the water. In contrast, the concentrations of radioactivity in the contacted portions of the other two pathways are always less than or equal to the corresponding concentrations in the water.

The five pathways are of varying importance at the different generic sites. The relative importance of the pathways depends both upon the concentrations of radionuclides in the pathways and upon the sizes of the exposed populations. Comparisons of the pathways for each type of generic site are presented in Table 4.3. (Both the pathways and their potentially exposed populations are discussed in more detail in Appendix E.)

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*If the uptake and elimination rates (or deposition and weathering rates) were known well enough, concentration factors (or distribution coefficients) would not be necessary (see Appendix C).

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Table 4.2 Times of Importance Along the Pathways to the Human Population

Pathway	Time to Reach Pathway(a)	Persistence in Pathway	Processing Delay
Drinking Water	0	depends on flushing time of waterbody	
Aquatic Foods	hours to years	hours to years(b)	hours to years
Finfish	days to years	weeks to years(b)	hours to months
Crustaceans	days to months	days to years(b)	hours to months
Mollusks	days to months	days to months(b)	hours to months
Kelp, Algae	hours to days	hours to days(b)	hours to years
Irrigated Foods	hours to years	years to centuries(b)	hours to months
Grains	days to months	years to centuries(b)	weeks to months
Vegetables	hours to months	years to centuries(b)	days to weeks
Animal By-Products	days to months	day to months(c)	days to weeks
Animal Products	days to years	months to years(c)	days to months
Shoreline	0 to months	years to centuries(b)	0
Swimming	0	depends upon flushing time of waterbody	0

[a] Measured from time of introduction of radionuclide into waterbody.
[b] Measured from time of removal of radionuclide from water by flushing.
[c] Measured from time of removal of contaminated crops from the animal's foodchain.

Table 4.3 Relative Importance of the Five Pathways to the Dose to the Human Population at the Generic Sites

Pathway Waterbody	Aquifer	Great Lake	Estuary	Ocean	River System
Drinking Water					
Concentration	high	low	-	-	moderate
Population	small	large	-	-	moderate
Importance[a]	1	1	-	-	1
Aquatic Foods					
Concentration	-	low	moderate	low	moderate
Population	-	large	large	large	moderate
Importance	-	2	1	1	2
Irrigated Foods					
Concentration	high	low	-	-	moderate
Population	small	small	-	-	small to large
Importance	2	4	-	-	4
Shoreline					
Concentration	-	low	moderate	low	moderate
Population	-	large	large	large	large
Importance	-	3	2	2	3
Immersion					
Concentration	-	low	moderate	low	moderate
Population	-	large	large	large	large
Importance	-	5	3	3	5

[a] Importance denotes relative importance of the resultant population dose for the pathway at the designated waterbody; the relative importance of the pathways can differ from the indicated order at specific sites.

For the five generic sites, substantial variations in the potential hydrospheric consequences of a meltdown accident result from the differences in both the concentrations of radioactivity in the waterbodies and the related populations-at-risk. Additional variations result from differences in the characteristics of the dominant pathways at the five types of sites; the largest of these variations are those due to two factors: the type of water (salt or fresh) and the basic type of hydrology (isolated aquifer or surface waterbody).

For a given type of hydrological site, sizeable site-to-site variations in the potential consequences are caused by differences in both the hydrological properties (especially flushing and dilution rates) and the associated populations-at-risk. Additional variations are caused by differences in two sets of factors: natural pathway characteristics (for example, the aquatic ecosystem affected and the flooding cycle undergone); and human-imposed pathway features (for example, the type of water treatment employed and the irrigation technique utilized).

Reference

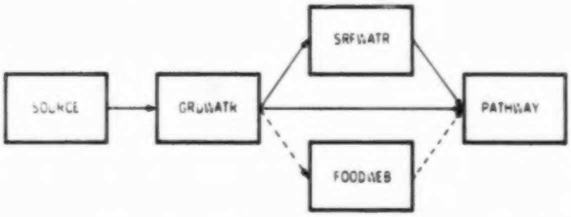
[1] E. V. Guisti and E. L. Meyer, "Water Consumption by Nuclear Power Plants and Some Hydrological Implications," *USGS Circulation 745*, USGS, 1977.

5 Models

5.1 Introduction

Until this point, the discussions have focused on reviewing the problems involved in considering the movement of radionuclides in the environment. This chapter outlines the modeling effort that has been performed for this study to consider the relevant aspects of these problems.

The models describing the amounts of radioactivity and the rates of the releases to the hydrosphere after a core-melt accident are discussed in Section 5.2. The groundwater and the surface-water transport models, which consider the movement through the hydrosphere, are outlined in Section 5.3. In the following section, 5.4, the models which describe the concentrations of radioactivity in the liquid exposure pathways are given. In the last section, 5.5, the dosimetry and the health effects models are reviewed. All the models and the parameters utilized in them are considered in much greater detail in Appendices A through E. A schematic flow draft for all of the related computer programs is presented in Figure 5.1 (also see Figure 1.2).



SOURCE: magnitudes and rates of releases to the hydrosphere
GRDWATR: movement of radionuclides through the ground
SRFWTR: movement of radionuclides within surface waterbodies and along exposure pathways
FOODWEB: transport within surface waterbodies and movement through the aquatic ecosystem
PATHWAY: dosimetry and health effects considerations

Figure 5.1. Schematic Outline of Calculational Procedure

5.2 Source

Three types of radioactive releases to the hydrosphere have been considered in this study: leaching from the melt debris; depressurization of the containment atmosphere; and escape of sumpwater from the containment. All aspects of the source term description are presented in greater detail in Appendix A.

5.2.1 Models

The depressurization and sumpwater releases have been taken to occur instantaneously at the time of meltdown. Leaching has been assumed to occur more gradually. Two initiation times have been considered for leaching: the time of the meltdown and one year after the start of meltdown. The latter corresponds to the prediction of the average time required for sufficient cooling to permit groundwater to contact the melt (Appendix A).

The leaching rate of any given radionuclide has been taken to be represented approximately by the expression $be^{-bt-\lambda t}$. In this expression, b denotes the fractional rate of release by leaching and λ is the radioactive decay constant.

5.2.2 Parameters

5.2.2.1 Rates

A range of rates have been used to consider releases by leaching from the melt debris. An empirically-based set of relative rates was used to order the rates for the relevant elements. The same information was also utilized to obtain a range of absolute rates appropriate for a variety of possible final melt compositions and properties. The relative and the absolute rates employed in the calculations are presented in Subsection A4.3 of Appendix A.

5.2.2.2 Magnitudes

To facilitate the comparison of the results of this study and those of the RSS, the magnitudes of the source terms for the various types of releases have been based primarily on the release fractions given in the RSS.[1,2] These have been combined with the core inventory of radionuclides used in that study.[1] All the volatilized radionuclides have been assumed to be found either in the airborne release or in the sumpwater release. The values employed for the magnitudes of the various types of releases are presented in Section A5 of Appendix A.

5.3 Hydrospheric Transport

In this report, five types of hydrological sites are considered: "dry" (isolated aquifer), large lake, estuary, ocean and river system. The first of these has only groundwater transport, whereas each of the last four has both groundwater and surface-water transport.

The general movement of a substance in any waterbody can be described by

$$\frac{dc}{dt} = -u_x\frac{dc}{dx} - u_y\frac{dc}{dy} - u_z\frac{dc}{dz} + \frac{d}{dx}\left[D_x\frac{dc}{dx}\right] + \frac{d}{dy}\left[D_y\frac{dc}{dy}\right] + \frac{d}{dz}\left[D_z\frac{dc}{dz}\right] - \lambda c + S(t), \qquad (5.3\text{-}1)$$

where c is the concentration of the material in the water; u_x, u_y, and u_z are the x, y and z components of the water velocity, respectively; D_x, D_y and D_z are the dispersion coefficients for the x, y and z directions, respectively; λ is the radioactive decay constant of the material; and S is the rate of the release (and/or removal) of the material into (and/or from) the water. The different terms in (5.3-1) are of varying importance in the different types of surface and subsurface waterbodies.

Solution of the above equation can be greatly simplified if the following approximations are utilized:

1. all the coefficients in Equation (5.3-1) are constants with respect to both spatial and temporal variations;
2. the waterbodies have straight boundaries and are of uniform depth and width, or else, the waterbodies are uniformly mixed; and
3. the waterbodies are isotropic with respect to dispersal of contamination.

In this study, these assumptions have been employed for aquifers and estuaries; they have also been employed sectionally for lakes, oceans, and river systems.

In general, analytical solutions of Equation (5.3-1) can be readily obtained for instantaneous releases (from idealized sources) into waterbodies described by the just-mentioned assumptions. In contrast, such analytical solutions are often not available for more realistic rates of release. However, for an arbitrary continuous release of radioactivity, the solution of (5.3-1) can be written as

$$f(t) = \int_0^t S(t')\, f'(t-t')\, dt', \qquad (5.3\text{-}2)$$

where f(t) is the solution at time t for the continuous release; S(t') is the source rate of continuous release at time t'; and f'(t-t') is the solution at time (t-t') for an instantaneous release. Thus, one can obtain straightforwardly concentrations for continuous releases by combining in Equation (5.3-2) the appropriate rates of releases with solutions for instantaneous releases. Such convolution techniques have been utilized throughout the modeling of transport through the hydrosphere.

For river systems, an alternative solution technique has also been employed. It will be discussed later. All of the groundwater and surface-water transport models are discussed in detail in Appendix B.

5.3.1 Groundwater Transport

Initially, the release to the hydrosphere is expected to be to the substratum beneath the containment. If human exposure is directly from contaminated aquifer water, then the concentration of the radioactivity at the point of withdrawal from the aquifer is important. However, if exposure is from contaminated surface water, then the amount of radioactivity entering the surface waterbody from the aquifer is relevant. Models for calculating the concentration at any point within the aquifer and for calculating the amount of material crossing the interface between the aquifer and the nearest surface waterbody are reviewed in this section. Both of the models used in this study to describe radionuclide transport in the ground are essentially those employed in the NRC Liquid Pathway Generic Study (NUREG-0440).[3]

5.3.1.1 Aquifer

5.3.1.1.1 Concentration Model

A one-dimensional transport model with dispersion in all directions has been used to describe the movement of any radionuclide mass through the ground. The equation which describes the movement in such a model can be written as

$$a\frac{dc}{dt} = u_x\frac{dc}{dx} + D_x\frac{d^2c}{dx^2} + D_y\frac{d^2c}{dy^2} + D_z\frac{d^2c}{dz^2} - \lambda ac + aS, \qquad (5.3\text{-}3)$$

where a is the retention factor of the radionuclide for the substratum under consideration and the other variables and parameters are as previously defined. In general, the retention factor a is approximately proportional to the distribution coefficient, K_d.*

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*For equilibrium conditions, K_d for a given radionuclide usually represents the ratio of the concentration of that radionuclide absorbed to the soil to the concentration of that nuclide in solution.

All the parameters in Equation (5.3-2) and in the related expressions have been assumed to be independent of location; that is, they have all been taken to be constants. Therefore, the solution for the concentration at any point due to an arbitrary release can be readily obtained as previously described.

5.3.1.1.2 Parameters

A survey of all the existing and many proposed LWR reactor sites was conducted to determine the ranges and the distributions of the important subsurface hydrological parameters which actually exist at reactor sites.[4] That is, an attempt was made to characterize a representative substratum for an LWR site. (See Subsection B2.2.1 in Appendix B.)

To characterize each reactor site, some of the available information was combined to estimate the average groundwater travel time from the site of each containment to the nearest surface waterbody of interest. Those sites which had groundwater travel times of more than 10^5 days (approximately 274 years) were defined to be dry sites. Only at such sites was direct exposure via aquifer usage considered.

All of the hydrological parameters employed in the calculations are given in Subsection B2.2.2. They were obtained primarily by the data-gathering effort just described. The K_d's used for all groundwater calculations in this study[5] are presented in Subsection B2.2.3.

5.3.1.2 Flow into Surface Waterbodies

5.3.1.2.1 Flux Model

The amount of a radionuclide reaching any surface waterbody is given by the flux across the aquifer-waterbody interface as a function of time. The flux of material passing across a plane (yz) perpendicular to the direction of flow (x) can be written as

$$F_x = Q\epsilon_e \iint \left(u_x c - D_x \frac{dc}{dx} \right) dy\, dz. \qquad (5.3\text{-}4)$$

Just as in the situation for the concentration, the flux across a face at a distance x from the source can be written in closed analytical form for an instantaneous release from certain shapes of sources (for example, see Appendix B, Equation (B2-16)). As a result, Equation (5.3-2) can be used for the flux due to a continuous release of radioactivity from the source, with f′(t-t′) being the flux for an instantaneous release from a source of the same shape.

5.3.1.2.2 Parameters

The hydrological parameters taken from the data survey already described in Subsection 5.3.1.1.2 were employed. Because of the extreme sensitivity of the liquid pathway results to certain aspects of the description of the transport of the radionuclides through the ground, a single representative site was thought to be inadequate to consider that transport. Instead, calculations were performed for an appropriate range of groundwater travel times for each type of surface-water site. The previously-mentioned set of distribution coefficients was used in all the groundwater calculations.

5.3.2 General Surface-Water Models

The four types of generic surface-water sites considered in this study are the following: large lake (Great Lakes), estuary, ocean and river system. Surface-water transport is modeled in two slightly different manners, depending upon the exposure pathway models under consideration. In general, transport in various surface waterbodies has been modeled in the manners that will be outlined in the first part of this subsection. However, for the time-dependent aquatic-food (foodweb) pathway model (see Subsection 5.4.2), the somewhat simplified surface-water transport models described in the last part of this subsection have been employed. The general surface-water transport models are discussed in greater detail in Appendix B, while the foodweb surface-water models are considered more completely in Appendix C.

All the general models used to describe surface-water transport and sediment-nuclide interactions are essentially those employed in the NRC Liquid Pathway Generic Study (NUREG-0440).[3] The modifications performed for this study and the differences in the ways in which they are utilized in the two studies are indicated in Appendix B.

In this study, the simplifying assumptions previously mentioned in Section 5.3 have been employed for modeling all surface waterbodies and the advantages of analytical solutions have been utilized. The general treatment for each of these surface waterbodies is analogous to that already presented for aquifers. For any waterbody residing in a series of other waterbodies, appropriate combinations of the spatially-varying solutions and mixed-tank descriptions have been used.

A river system can be modeled in the same manner as the other types of surface waterbodies have been. However, the advantage of analytic solutions can be lost in dealing with such a complex system. To adequately describe all the variations along a typical river system, another model has also been developed. In that model, the river is divided into a series of sections with variation of certain important coefficients being permitted between those sections. The resulting series of equations has been solved by employing methods typically utilized for dealing with differential equations.

5.3.2.1 Great Lakes

5.3.2.1.1 Great Lakes Models

Three models are used in this study to consider movement of radionuclides within any Great Lake: nearshore, surface mixed-tank, and complete mixed-tank. The first describes local movement along the coast; the second describes mixing in the upper 50 to 75 feet of the lake; the third describes the mixing in the complete lake due to seasonal turnover. A fourth model is used to consider the movement of radionuclides into successive waterbodies: a connected series of completely-mixed tanks. This last model primarily describes movement into and within the other Great Lakes. Sediment effects are considered in all the models except the nearshore region model.

The concentration of a radionuclide in the coastal currents in the local or nearshore region can be described by

$$\frac{dc}{dt} = -u_x\frac{dc}{dx} + E_x\frac{d^2c}{dx^2} + E_y\frac{d^2c}{dy^2} - \lambda c + S(t), \qquad (5.3\text{-}5)$$

where E_x and E_y are the alongshore and the offshore dispersion coefficients, respectively, and the other parameters are as described previously. The conditions given in Section 5.3, other than the uniform-mixing one, can be assumed to consider the concentration close to the shore. For a lake, as for any other surface waterbody initially contaminated by the accident, the source term essentially represents the appropriate amount of material passing across the aquifer-waterbody interface as a function of time.

In a zeroth-order approximation, each of the Great Lakes can be described as a completely-mixed reservoir. Thus, the concentration for the mth lake in a series of such reservoirs can be described by

$$\frac{dc_m}{dt} = \frac{q_{m-1}}{V_{m-1}}c_{m-1} - \lambda_i c_m - \frac{q_m}{V_m}c_m, \qquad (5.3\text{-}6)$$

where V_i is the volume of the ith lake and q_i is the flow rate of water out of that lake. The concentrations due to any arbitrary release can be obtained trivially. The source of radioactivity for the first mixed tank is the flux of material passing across the interface between the initially contaminated nearshore region and the rest of the lake; likewise, the source for the next lake is just the flux of material passing into it as a result of flushing of the first lake.

Simultaneously with transport by surface-water movements, the radionuclides undergo interactions with the sediment. In the sediment-radionuclide interaction model for lakes,[3,6] the radionuclides on the suspended sediment particles are assumed to be in equilibrium with respect to the nuclides in the water; that is, the concentration of a radionuclide residing on the suspended sediment is taken to be equal to the product of the appropriate distribution coefficient and the concentration of the nuclide in the water. Therefore, as sedimentation occurs, the radionuclides residing on the sediment particles are removed from the water column at a rate which is proportional to the net sedimentation rate, the appropriate distribution coefficient and the concentration in the water.

In a similar fashion, the nuclides in the interstitial water of the deposited sediment are assumed to be in equilibrium with any nuclides adsorbed to that sediment. However, the concentration in the interstitial water can be different than the concentration in the water column. Therefore, a net diffusion of radionuclides can occur across the sediment-water interface. The net rate of flow from the interstitial water to the water column is assumed to be proportional both to the difference in the concentrations in the water in the sediment interstices and the water in the water column and to the effective diffusion coefficient.[3,6-8] In this report, the diffusional transfer rate is taken to be element independent.

If sedimentation occurs at a fast enough rate, radionuclides can be buried and lost to further contact with the water. The rate of loss by such burial is proportional to both the sedimentation rate and the concentration of radioactivity in the deposited sediment. In this report, the burial depth is taken to be element independent.

The models just discussed can be combined so that the effects of sediment-water interactions can also be considered for series of reservoirs or lakes. Such models have also been utilized.

5.3.2.1.2 Parameters

The hydrological parameters for the Great Lakes are presented in Subsection B3.1.2.1. The parameters for Lake Michigan were employed in the calculations for the generic large lake. The distribution coefficients utilized in the calculations are considered in Subsection B3.1.2.2. Their values are thought to be appropriate for considering some surface bodies of fresh water.

5.3.2.2 Estuaries

5.3.2.2.1 Estuary Model

A one-dimensional sectionally-averaged model is employed in this study to consider all estuaries. Tidal influences are taken to be adequately described by a "tidally-averaged" approximation in which the effects of tidal oscillations are taken to be represented by increased longitudinal dispersion. In such a model, the concentration of a radionuclide resulting from the net downstream flow and the accompanying tidally-induced dispersion can be given by

$$\frac{dc}{dt} = -u_x\frac{dc}{dx} + E_x\frac{d^2c}{dx^2} - \lambda c + S(t), \qquad (5.3\text{-}7)$$

where E_x is the tidally-averaged longitudinal dispersion coefficient.

Simultaneously with the transport by surface-water movements, the radionuclides undergo interaction with the sediment. In the estuary sediment-radionuclide interaction model, the radionuclides residing on the suspended sediment particles are assumed to be in equilibrium with respect to the radionuclides dissolved in the water column. Furthermore, the nuclides residing within a certain top thickness of deposited sediment are also assumed to be in equilibrium with the nuclides in the water column. This thickness is taken to be composed of the sediment which is frequently resuspended.

As sedimentation occurs, the radionuclides residing on the lower sediment particles of the upper sediment layer are taken to be removed from further interaction by burial. Diffusion of radionuclides back into the water column is ignored in this estuary model.

The equation describing the resulting concentration of any radionuclide in the water phase is like Equation (5.3-7), except that the magnitudes of the velocity and the dispersion coefficient are both modified to take sediment-radionuclide interactions into account. The values of both are decreased as the level of the interaction is increased.

5.3.2.2.2 Parameters

The hydrological parameters employed for the calculations of two generic estuaries are presented in Subsection B.3.2.2.1. These parameters are for two Middle-Atlantic estuaries. The sediment distribution coefficients employed for estuaries are discussed in B3.2.2.2. The utilized coefficients are some considered to be appropriate for saltwater conditions.

5.3.2.3 Oceans

5.3.2.3.1 Ocean Models

The concentration of a radionuclide in the nearshore coastal currents can be described by

$$\frac{dc}{dt} = -u_x\frac{dc}{dx} + E_x\frac{d^2c}{dx^2} + E_y\frac{d^2c}{dx^2} - \lambda c + S(t), \qquad (5.3\text{-}8)$$

where E_x and E_y are the alongshore and the offshore dispersion coefficients, respectively. Under the assumptions given in Section 5.3, other than the uniform-mixing one, the concentration is given by the same expression as for the nearshore region in a Great Lake.

The offshore region of the continental shelf is taken to be described by a mixed tank. The concentration in that region is given by the same expression as for the mixed region in a Great Lake. Flow out of the tank is taken to be into the deep portions of the ocean. Inasmuch as the flushing times on the continental shelves are relatively short and the sedimentation rates are generally low, the effects of sedimentation are ignored.

5.3.2.3.2 Parameters

Parameters believed to be characteristic of an open Atlantic coastal site have been utilized in the calculations for a generic ocean site. The hydrological parameters employed in the calculations are presented in Subsection B3.3.2.1.

5.3.2.4 River Systems

5.3.2.4.1 River System Models

A river system is modeled as a series of compartments, with each representing either a free-flowing stretch or a reservoir section. The concentration of any radionuclide in any section of the system can be written as

$$\frac{dc}{dt} = -u\frac{dc}{dx} + \frac{d}{dx}\left(E_x\frac{dc}{dx}\right) - \lambda c + S(t). \qquad (5.3\text{-}9)$$

The concentration is taken to include the radioactive material dissolved in the water plus that sorbed to sediments traveling with the water. It is assumed that the longitudinal velocity of sediments carried by the water is equal to the water velocity. The descriptions of sediment behavior and sediment-radionuclide interactions employed for the river system model are identical to those employed for the large lake model (Subsection 5.3.2.1.1).

Two methods of solution have been employed: the integral method previously considered for the other waterbodies and a method using differential equation techniques. In the latter, the equations for the concentrations in the water and in the sediment have been approximated by a combination of finite difference equations. The equations have been simplified for reservoirs, each of which is treated as either a single mixed tank or a series of mixed tanks, and for dilution points (tributaries), in which only dilution and transport occur.

5.3.2.4.2 Parameters

Two specific river systems have been considered in the calculations: the Tennessee-Ohio-Mississippi and the Missouri-Mississippi. The hydrological parameters utilized are presented in Subsection B3.4.2.1.6 and the distribution coefficients are discussed in Subsection B3.4.2.2. The distribution coefficients employed are some thought to be adequate for certain freshwater bodies.

5.3.2.5 Series of Waterbodies

The complete impact of an accidental release of radioactivity into any waterbody is not necessarily determined by considering just the effect within that waterbody. The water from most waterbodies flows into other waterbodies. The modeling of flow through the series of Great Lakes and down river systems, including and excluding reservoirs, has already been reviewed.

For both the river and the estuary models, the flux of material passing out through the mouths of these bodies has been calculated using

$$F_x = \iint \left(u_x c - D_x \frac{dc}{dx} \right) dy\, dz, \qquad (5.3\text{-}10)$$

where u_x is the magnitude of the net velocity of the isotope under consideration at the point of efflux. When it was determined that a sufficiently large flux of radioactivity had passed through the mouth of a river or estuary, then the efflux of that waterbody was used as input to the next waterbody in the series. No calculations were performed for impacts due to the residence of the radioactivity in the deep portions of the ocean.

5.3.3 Foodweb Surface Water Models

The surface-water models employed in the time-dependent aquatic food (foodweb) model are somewhat less complicated than most of the models just described. For all waterbodies, the foodweb surface-water models consist of two mixed tanks in series. A Great Lake is divided into a nearshore region and the rest of the lake. An estuary is divided into two regions, one downstream from the other; the downstream region can represent either an estuarial region or a nearshore ocean region. An ocean is divided into a nearshore region and a continental shelf region. In all cases, the relevant equations are basically the same as those considered in Subsection 5.3.2.1.1 for a series of lakes with sediment-radionuclide interactions. However, the modifications of the sediment-nuclide interaction model previously mentioned for estuaries and oceans are used for those waterbodies.

5.4 Movement Along Pathways

The five types of exposure pathways considered in this study are the following: ingestion of drinking water, ingestion of aquatic foods, ingestion of irrigated foods, shoreline usage and immersion in water. All of the pathway models, other than the foodweb model, are discussed in detail in Appendix D. The foodweb model is considered in Appendix C and in a separate report, SAND80-1670 (NUREG/CR-1597).

5.4.1 Pathway Models

5.4.1.1 General Models

The pathway models estimate the concentrations of the radionuclides in the portions of the exposure pathways contacted by humans. With the exception of the formulae used for shoreline exposure, for drinking water ingestion and for aquatic food ingestion, the expressions utilized for estimating the concentrations in each of the five pathways are essentially adapted versions of those presented in Regulatory Guide 1.109.[9] The expression for the shoreline concentration has been additionally modified to include the effects of some weathering. The model for considering drinking water concentrations has been altered to include the effects of sediment removal during water treatment. The aquatic food concentrations have been considered with three different models: a time-independent model, analogous to that presented in Regulatory Guide 1.109;[9] a one-compartment time-dependent model, analogous to that utilized in

NUREG-0440;[10] and a multi-compartment time-dependent foodweb model, developed for this study. The two more complex aquatic food models differ from the Regulatory Guide 1.109 one by their inclusion of the interactions of the aquatic organisms with the environment at varying higher levels of sophistication.

The Regulatory Guide 1.109 models for all the pathways are essentially steady-state models in which the concentration of any radionuclide in the water is taken to be a constant and the corresponding concentrations in the contacted portions of the pathways are taken to be simple multiples of the concentrations in the water. The extensions of those models employed here permit variations of the pathway radionuclide concentrations resulting from changes in the concentrations in the water with time. The modifications used here even allow for some time-dependent accumulation in and elimination from the relevant pathway components in those three pathways which involve extensive interactions of the radionuclides with portions of the environment. In particular, the concentration of radionuclide i in any one of the three more complicated pathways has been taken to be of the form

$$C_{ip}(t) \simeq \ldots e^{-\lambda_{ei}t} \int_{o}^{t} \ldots \; C_{iw}(t')\, e^{+\lambda_{ei}t'} \, dt', \qquad (5.4\text{-}1)$$

where $C_{iw}(t')$ is the concentration of the radionuclide in the water, and λ_{ei} is an effective removal rate from the pathway. The exact forms of these expressions are given in Appendix D.

The complexity of the foodweb model permits consideration of the variations of the radionuclide concentrations in all parts of the aquatic ecosystem and not just in the contacted portions of the pathway as is done in the other improved aquatic food model. This model is outlined in Subsection 5.4.2.

5.4.1.2 General Parameters

The values for all of the parameters in the various pathway models, other than the foodweb model, are presented in Sections D2.1 and D2.4 of Appendix D. The pathway model parameters, other than those involving the time-dependent behavior, have been taken mostly from Regulatory Guide 1.109. Those involving time-dependent behavior have been estimated from various sources for this study.

5.4.2 Foodweb Model

In the foodweb model, the buildup of the radioactivity in all the aquatic organisms is taken to be a function not only of time but also of the interconnections within the appropriate foodweb. This is unlike the other pathway models, in which the descriptions of the movement of the radionuclides within the pathways to the human population are taken to be linear, essentially along a chain.

5.4.2.1 Model

In the foodweb model, the surface waterbody under consideration is divided into two (or more) spatially-distinct regions, in each of which a uniform mixing approximation is made for the water. The environment within each region is divided into an appropriate set of interacting physical, detrital, and biological compartments, examples of which might be deposited sediment, suspended detritus, and phytoplankton, respectively. The masses in each of these compartments are assumed to be fixed; radioactivity is conserved. The movement of radionuclides among the various compartments is examined as a function of time.

The transfer of a given radionuclide among the various compartments is taken to be described by the following set of equations:

$$\frac{dx_j}{dt} = -\sum_{i \neq j} \lambda'_{ji} x_j + \sum_{i \neq j} \lambda_{ij} \, \Delta x_j + \sum_{k} \lambda_{kj} \Delta x_j - \lambda_j^e x_j \;,$$

$$j = \text{biological}; \qquad (5.4\text{-}2)$$

$$\frac{dx_j}{dt} = -\sum_{i} \lambda_{ji} \Delta x_i + \sum_{k \neq j} \lambda_{kj} x_k - \sum_{k \neq j} \lambda_{jk} x_j \;,$$

$$j = \text{physical},$$

$$j \neq \text{detritius}; \qquad (5.4\text{-}3)$$

$$\frac{dx_j}{dt} = -\sum_{i} \lambda_{ij} \Delta x_i + \sum_{i} \lambda'_{ji} x_i + \sum_{k \neq j} \lambda_{kj} x_k - \sum_{k \neq j} \lambda_{jk} x_j + \sum_{i} \sum_{i' \neq i} \lambda'_{ii'} x_i - \sum_{i} \sum_{i \neq i} \lambda_{ii'} \Delta x_i + \sum_{i} \lambda_i^e x_i$$

$$j = \text{detritus}\;; \qquad (5.4\text{-}4)$$

where x_j denotes the amount of a given radionuclide in compartment j, Δx_j represents essentially the difference between the amount of a radionuclide actually in biological compartment j and the amount that would be present if the compartment contained the amount predicted by the appropriate concentration factor, λ_{ij} (or λ'_{ij}) denotes the fractional rate of transfer of activity from compartment i to compartment j, and λ^e_i represents the composite elimination rate for compartment i due to death and metabolic elimination. The summations with respect to i and i' are over all biological compartments; the summations with respect to k are over all physical and detrital compartments.

The terms in the equations are taken to represent the significant transfers between the compartments involved. For example, in the general equation for any biological compartment, the first summation denotes the losses due to ingestion by other compartments, the second denotes the gains by assimilation after ingestion of other biological compartments, the third denotes the assimilation from intake of portions of physical compartments, and the fourth represents losses due to metabolic elimination and death.

The transfers of activity among the water and the sediment compartments are considered by using basically the same models previously described in Subsection 5.3.2. The only fundamental differences are that in the foodweb model for either a Great Lake or an ocean, the nearshore region is taken to be uniformly mixed and for an estuary, the entire estuary is taken to be a series of completely-mixed tanks.

The time-dependent transfers of activity among the biological compartments are considered to be due to ingestion and assimilation. The amount of activity ingested by compartment j due to feeding on compartment i is taken to be equal to the product of the mass of i ingested by j times the concentration of radioactivity in i. The amount of activity actually assimilated by compartment j due to feeding on i is taken to be the product of three factors: the amount of activity in i ingested by j, the appropriate assimilation fraction, and a weighting factor which is proportional to the "distance" of compartment j from concentration-factor "equilibrium." The radioactive material which is ingested from the biological compartments but which is not assimilated is assumed to go to the detrital compartments.

5.4.2.2 Foodweb Parameters

Three specific "sites" have been investigated with the foodweb model: a Lake Michigan site, a Middle Atlantic estuary site, and a southern California Pacific Ocean site. Biological and radiobiological data bases have been developed for each of these sites.[11] The values of the parameters employed at each of the three sites are presented in SAND80-1670 (NUREG/CR-1597).

5.5 Dosimetry and Health Effects

The radiation doses received via the liquid exposure pathways by both individuals and by the entire human population are considered. In addition, the number of resultant adverse health effects are estimated. All of the dosimetry and health effects models are described in detail in Appendix D.

5.5.1 Dosimetry Models

5.5.1.1 Individual Dose

5.5.1.1.1 General Expression

In general, the dose received by an individual between times 0 and t by means of pathway p can be written as

$$\text{dose}_p = U_p \sum_i d_{ip} f_{ip} e^{-\lambda_i t_p} \int_0^t C_{ip}(t')\,dt' , \qquad (5.5\text{-}1)$$

where U_p is the individual usage rate, i denotes the isotope, d_{ip} is the appropriate dose factor, f_{ip} is a processing or other modification factor, λ_i is the radioactive decay constant, t_p is the transportation and/or processing delay time, and $C_{ip}(t)$ is the concentration of the isotope in the pathway. The factor $e^{\lambda_i t_p}$ allows for decay during the period after removal from the waterbody and before exposure.

5.5.1.1.2 Individual Parameters

The ingestion dose factors employed have been generated especially for this study at ORNL[12] using the same dosimetry codes that were employed for the RSS. The external dose factors utilized for shoreline and immersion exposures have been taken from the "Year Two-Thousand Study."[13] The other parameters have been obtained primarily from Regulatory Guide 1.109.

5.5.1.2 Population Dose

5.5.1.2.1 General Expression

The population can be regarded as being composed of subgroups differing in either usage patterns or concentrations experienced in the pathway or both. As a result, the dose received by a population from exposure to pathway p can be written as

$$\text{dose}_p = \sum_{p'} P_{pp'}\, U_{p'} \sum_i d_{ip} f_{ip} e^{-\lambda_i t_p} \int_0^t C_{ipp'}(t')\,dt' , \qquad (5.5\text{-}2)$$

where $P_{pp'}$ is the population size of subgroup p' in pathway p, and all other symbols are as previously defined.

5.5.1.2.2 Population Parameters

A uniform usage rate has been assumed for all members of a population for each pathway. Furthermore, the products of the population sizes and the individual usage rates have been taken to remain constant over the entire time period under consideration. Uniform spatial distribution of affected populations has been assumed for all five types of hydrological sites.

The values used for the population sizes and densities and their usage rates are presented in Appendix E. Generic populations have been chosen by employing the numbers of reactors at various types of waterbodies as conceptual weighting factors. As a result, the following populations have been taken to be representative: Lake Michigan's for a large lake; both Chesapeake and Delaware Estuaries' for an estuary; the Atlantic continental shelf's for an ocean; and both the Tennessee-Ohio-Mississippi and the Missouri-Mississippi Rivers' for a river system. Averages of the actual dry site populations have been used for considering dry sites. The population data appropriate for the representative hydrological sites have been taken from various sources. All other parameters for Equation (5.5-2) are given in Appendix D.

5.5.2 Health Effects Models

Health effects models translate the radiation dose received by the human population into the resultant health effects such as cancers, thyroid effects and genetic mutations. To facilitate the comparison between the results of this study and the RSS, the health effects models used here are essentially identical to those used in the RSS.[1] They are discussed in more detail in Appendix D.

5.5.2.1 Acute Effects

In the RSS,[1] health effects were divided into acute, latent and chronic. The acute effects were due primarily to large early total-body doses of 100 rads or more to individuals. The acute effects considered were all somatic and included direct radiation damage to certain organs, thyroid impairment, prenatal and neonatal deaths, and sterility.

5.5.2.1.1 Acute Models

For acute effects, dose-response curves were employed to convert individual doses to health effects. Such curves plot individual dose versus the probability of the effect in an individual. Each of these curves is characterized by a low dose threshold below which no effect occurs and a high dose threshold above which the effect always occurs.

The total number of acute effects of type j can be written as

$$(\text{health effect})_j = \sum_{p'} P_{p'} \cdot \beta_j(\text{dose}_{p'}) \, , \qquad (5.5\text{-}3)$$

where $P_{p'}$ is the population for subgroup p', and $\beta_j(\text{dose}_{p'})$ is the probability of effect j for total individual dose p'. The total individual dose is the sum of the contributions from all possible pathways, $\{p\}$.

5.5.2.1.2 Parameters

The dose-response curves used to consider acute effects are presented in Appendix VI of the RSS.[1]

5.5.2.2 Latent and Chronic Effects

In the RSS,[1] the latent and the chronic effects were considered to be random phenomena based on the total doses to the entire population. In general, the doses to the individuals which resulted in latent and chronic effects were much less than those which caused acute effects. The effects included cancers, genetic effects and thyroid nodules, all occurring many years after the initial exposure. Most, if not all, of the effects obtained through liquid pathways are expected to be chronic.

In general, the total number of latent or chronic effects of type j can be written as

$$(\text{health effect})_j = h_j \cdot \sum_{p'} P_{p'} \, \delta_j(\text{dose}_p) \sum_{p} (\text{individual dose})_{pp'} \, , \qquad (5.5\text{-}4)$$

where h_j is the health effects factor for effect j (expected effects/population dose), $P_{p'}$ is the population for subgroup p', and $\delta_j(\text{dose}_{p'})$ is the dose effectiveness factor for individual dose p'. The various latent and chronic effects models differ in the values assumed for the health effects factors and the dose effectiveness factors.

5.5.2.2.1 Cancer Models

For latent and chronic cancer, an absolute-risk model was employed. That is, the risk of any cancer was assumed to be constant over a given finite plateau period, with no effects occurring before or after the plateau period.

Three different estimates were made of the latent and chronic cancers for almost all considered organs:

an upper bound, a central estimate and a lower bound. The upper bound was based on the BEIR or linear, no-threshold hypothesis.[14] In that model, it is assumed that the number of health effects varies directly with the population dose. No individual dose thresholds are employed.

The central estimate is based, in part, on the BEIR hypothesis but assumes that doses administered at lower dose rates or in lower total individual doses are less effective in causing most health effects. Dose effectiveness factors less than one are used to modify the effects calculated for individual low dose rates and/or low total doses.

The lower estimate assumes that doses received at very low dose rates and/or in very low individual doses are not effective at all in causing health effects. Consequently, an individual threshold dose rate or threshold total dose is assumed, below which no effect is considered to occur.

5.5.2.2.2 Cancer Parameters

Because of problems in applying the individual thresholds, the upper bound has been used for most calculations. The health effects factors for the types of cancer considered are presented in D3.3.2. The values used for these factors include credits for current survival and cure rates. More detail about the models is presented in D3.3.2 and in Appendix VI of the RSS.[1]

5.5.2.2.3 Thyroid Effects Models

The latent and chronic thyroid effects considered are the occurrence of thyroid nodules and the appearance of hypothyroidism. In discussing thyroid neoplasms, a linear no-threshold model has been employed; for hypothyroidism, a linear model with a lower dose threshold has been utilized.

5.5.2.2.4 Thyroid Effects Parameters

The health effects factors for the types of thyroid effects considered are given in D3.3.3. More detail about the models is presented in Appendix VI of the RSS.[1]

5.5.2.2.5 Genetic Effects

Genetic effects result from radiation of the chromosomal matter of the germ cells. The effects considered include single-gene disorders, multifactorial disorders and chromosomal irregularities. The damage is assumed to be entirely from irradiation of the fathers. The health effects factors employed for genetic effects were developed from the BEIR report.

5.5.2.2.6 Genetic Effects Parameters

The health effects factors for genetic effects are presented in D3.3.4. More detail about the genetic effects models is given in Appendix VI of the RSS.[1]

5.6 Limitations

The limitations of both the models and the parameters utilized in performing this study are considered in detail in the appendices of this report.

References

[1] U.S. Nuclear Regulatory Commission, *Reactory Safety Study, An Assessment of Risks in U.S. Commercial Nuclear Power Plants*, Appendix VI, 1975.

[2] U.S. Nuclear Regulatory Commission, Reactor Safety Study, *An Assessment of Risks in U.S. Commercial Nuclear Power Plants*, Appendix VII, 1975.

[3] U.S. Nuclear Regulatory Commission, "Liquid Pathway Generic Study," NUREG-0440, USNRC, Appendix B, 1978.

[4] S. J. Niemczyk, *A Summary of Hydrological Information for Nuclear Power Plants*, in preparation.

[5] H. C. Burkholder, M. O. Cloninger, D. A. Baker, and G. Jansen, "Incentives for Partitioning High-Level Waste", Nuclear Technology *31*, 202-217, 1976.

[6] R. S. Booth, "A Systems Analysis Model for Calculating Radionuclide Transport Between Receiving Waters and Bottom Sediments," in *Environmental Toxicity of Aquatic Radionuclides: Models and Mechanisms*, edited by M. W. Miller and J. N. Stannard, Ann Arbor Science Publishers, Ann Arbor, Michigan, 1976.

[7] A. Lerman and G. J. Brunskill, "Migration of Major Constituents from Lake Sediments into Lake Water and Its Bearing on Lake Water Composition," Limnology and Oceanography *16*, 880-890, 1971.

[8] F. T. Manheim, "The Diffusions of Ions in Unconsolidated Sediments" Earth and Planetary Science Letters 9, 307-309, 1970.

[9] U.S. Nuclear Regulatory Commission, "Calculation of Annual Doses to Man from Routine Releases of Reactor Effluents for the Purpose of Evaluating Compliance with 10 CFR Part 50, Appendix I," Regulatory Guide 1.109, Revision 1, USNRC, 1977.

[10] U.S. Nuclear Regulatory Commission, "Liquid Pathway Generic Study," NUREG-0440, USNRC, Appendix C, 1978.

[11] G. E. Runkle, "A Biological Survey of the Aquatic Foodweb at Lake Michigan, the Chesapeake Bay and the Southern California Ocean," in preparation.

[12] E. Eckerman, ORNL, to S. J. Niemczyk, Sandia Laboratories, private communication, 1979.

[13] J. F. Fletcher, W. L. Dotson, et al., "HERMES: A Digital Computer Code for Estimating Regional Radiological Effects from the Nuclear Power Industry," HEDL-TME-71-168, Hanford Engineering, 1971.

[14] National Academy of Sciences-National Research Council, *The Effects on Populations of Exposure to Low Levels of Ionizing Radiation, Report of the Advistory Committee on the Biological Effects of Ionizing Radiations*, BEIR Report, National Academy of Sciences, Washington, D.C., 1972.

6 Results for Situations Without Mitigating Actions

6.1 Introduction

The risk from radiation received via liquid pathways after a meltdown accident is generally thought to be small because of the potential of interdicting* the radioactivity on its way to the human population. However, such mitigating procedures may not always be possible. And even if they are feasible, the overall cost may preclude such actions from being taken. In this chapter, the consequences of possible releases to the hydrosphere are considered, assuming that either interdictive procedures are not taken or else, if they are taken, they are not successful.

The radiation dose received by the human population and the resultant health effects are determined by many factors. This chapter discusses these consequences as functions of the more important of the factors. It also attempts to place these consequences and their related risks into a relevant framework so that they can be viewed from a useful perspective.

In Sections 6.2 through 6.4 of this chapter, the consequences and the dominant factors affecting them are discussed. In Section 6.2, the radiation dose received by the entire human population is considered. First, that dose is examined as a function of both the accident type and the rate(s) of release. Next, the population dose is considered as a function of the subsurface hydrology of the accident site. Then, the dependence of the radiation hazard on relevant surface waterbody characteristics and on dominant exposure pathway properties is investigated. After that, the population dose is described in terms of the potential site-to-site variations which exist at actual reactor sites. In Section 6.3, the radiation dose received by individuals in the exposed population is considered. The individual dose is examined in less detail than the population dose and is investigated as a function of only two factors: characteristics of the closest surface waterbody(ies) and properties of the dominant exposure pathways. In Section 6.4, the resultant health effects are reviewed briefly.

In the next several sections of this chapter, Sections 6.5 through 6.7, the interpretation of the results considered in the earlier sections is discussed. In Section 6.5, the risks of certain consequences resulting from releases to the hydrosphere are compared to those of comparable consequences from releases to the atmosphere.[1] In Section 6.6, the effects of some assumptions used in the study on the estimated consequences are explored; in particular, the extent to which certain assumptions are conservative is addressed. In Section 6.7, some limitations of this study, as they relate to interpretation of the results, are reviewed.

In the last section of this chapter, Section 6.8, the results are reviewed. In particular, the consequences of the liquid pathways resulting from the assumption that effective interdictive measures are not instituted are summarized. Details of all the results reviewed in this chapter are presented in Appendix G.

6.2 Population Dose

Potential radiation dose received by the entire human population can be used as one measure of the risk of a meltdown accident.*

6.2.1 Population Dose as a Function of Accident Type and Release Rate

6.2.1.1 Magnitude of Release

The total radiation dose resulting from a given radionuclide would be essentially directly proportional to the total amount of that nuclide which reached accessible portions of the hydrosphere. This amount would be, in turn, dependent upon the total quantity of the radionuclide which were released into both accessible and inaccessible regions of the hydrosphere.

The overall radioactive content of the melt debris, the sumpwater, and the containment atmosphere would each depend upon the particular accident scenario which were followed, as well as the design, the size, and the status of the core of the reactor which underwent meltdown. Unlike the situation for atmospheric releases, the most probable PWR meltdown category would result in the largest release of radioactivity to the hydrosphere. (In the atmospheric case, the most probable category would result in one of the smaller releases of activity to the atmosphere.[1]) The radioactive content of the melt debris would be approximately the same for all accident categories. However, the radioactive composition of both the sumpwater and the containment atmosphere would be highly variable. Only the melt debris and the sumpwater could contain substantial fractions of the core's initial radioactivity.*

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*In this study, the term interdiction is taken to encompass all actions, other than immediate emergency ones, which might be taken to mitigate the radiation exposure received by the human population and other inhabitants of the environment. This definition of interdiction departs from that used in the RSS. In that study, interdiction was taken to mean only the denial of the use of land and its improvements for a period of time to mitigate the radiation exposure.

*Only radiation doses and other consequences resulting from exposure within 100 years after the accident are explicitly considered in the results presented in this report.

The resultant population dose is considered in Table 6.1 as a function of the accident category for some illustrative examples. As expected, the estimated population doses from the melt debris releases are essentially the same for all the accident categories considered whereas the doses from the sumpwater releases are much more dependent on the accident category.

For a given accident category, the estimated population dose is related not only to the types and the initial magnitudes of any releases to the hydrosphere, but also to the fraction of the radioactivity which reaches the human population. This is, in turn, a strong function of the time it takes the radioactivity to travel to open portions of the environment. This latter factor depends significantly on the groundwater travel time to the nearest surface waterbody (or well) at the site. This dependence is considered in detail in Subsection 6.2.2 and its basic features are displayed in Figure 6.1.**

In that figure, the population dose is shown for a large lake site as a function of both type of release and groundwater travel time (t_{H_2O}) for instantaneous injection of the radioactivity in both the melt debris and the sumpwater into the groundwater. As can be seen, the predicted contribution of the sumpwater to the dose vanishes more rapidly than that of the melt debris as the groundwater travel time increases; this is, in part, because the former release is composed primarily of relatively short-lived radionuclides, whereas the latter contains many comparatively long-lived radionuclides.†

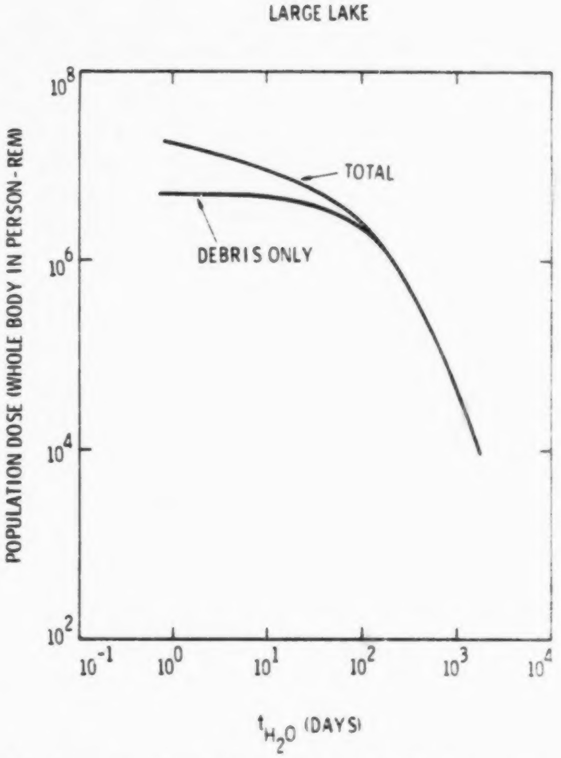

Figure 6.1 Population Dose for a Large Lake Resulting from Instantaneous Leaching of the Melt Debris and Instantaneous Entrainment of Sumpwater into the Groundwater after a PWR-7 Type Accident.

6.2.1.2 Rate of Release

The rate of release of radioactive material to the hydrosphere is expected to depend upon the type(s) of release(s), the accident scenario and certain characteristics of the site. In general, the passage of sumpwater and depressurized containment gases through the hole in the containment floor (formed by the melt) would be relatively rapid. The rates would depend largely upon the subsurface conditions.

In contrast to sumpwater and containment gas releases, leaching of at least some of the melt debris would generally be relatively slow. The rates of leaching of the melt debris would depend upon at least the following several factors: the factors determining the groundwater contact time for the melt debris; the chemical and physical properties of the debris; the groundwater flow characteristics of the site; and the chemical states of the radionuclides under consideration. The effect of leaching rates on the calculated population dose is considered in more detail in Subsection 6.2.4 of this chapter.

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*Although a significant portion of the core's radioactivity could be released directly to the atmosphere outside the containment, only a small fraction could be released directly to the hydrosphere by depressurization.

**Because the groundwater travel time is one of the most important parameters in estimating the potential consequences at a given site and because it cannot be treated realistically by using a single generic value, the dependence of the consequences on this time is considered throughout this chapter.

†It is also due to the slower movement of the Cs (one of the dominant contributors to the sumpwater dose) in the soil than Sr (usually one of the dominant contributors to the melt-debris dose).

Table 6.1 Estimated Population Dose as a Function of Accident Category[a,b]

	Estimated Population Dose (Whole Body in Person-Rem)							
	PWR-7		PWR-5		PWR-3		PWR-1	
Waterbody	Debris	Sump	Debris	Sump	Debris	Sump	Debris	Sump
Large Lake	5.6+6[c]	8.0+6	5.6+6	8.0+6	5.6+6	8.0+6	5.3+6	5.0+6
Nearshore	6.7+5	1.4+6	6.7+5	1.4+6	6.7+5	1.1+6	5.8+5	8.0+5
Small Estuary	2.4+7	4.4+7	2.4+7	4.4+7	2.3+7	3.5+7	1.6+7	2.5+7
Large Estuary	1.6+7	1.7+7	1.6+7	1.7+7	1.6+7	1.3+7	1.2+7	1.0+7
Ocean	2.6+6	1.3+6	2.2+6	1.3+6	2.1+6	1.0+6	1.7+6	7.0+5
Nearshore	2.6+6	1.3+6	2.2+6	1.3+6	2.1+6	1.0+6	1.7+6	7.0+5
Free-Flowing River	1.3+7	4.9+7	1.3+7	4.9+7	1.3+7	3.9+7	1.2+7	2.8+7
Dammed River	2.3+7	5.2+7	2.3+7	5.2+7	2.3+7	3.9+7	2.0+7	2.9+7

[a] A groundwater travel time of 1 day is assumed. Both debris and sumpwater releases are taken to occur essentially instantly.

[b] The corresponding doses from groundwater travel times other than 1 day are presented in Tables G10-G13 of Appendix G.

[c] 5.6+6 denotes 5.6 x 10^6.

In general, if the release times were short relative to the total time that the dominant radionuclides spent in the ground, then the exact release rates would be unimportant in determining the resultant population dose. However, if the release times were long in comparison to both the travel time in the ground and the decay times of the dominant radionuclides, then the exact release rates could be significant in estimating the dose. Because the sumpwater and the depressurization releases would tend to be dominated by radionuclides with shorter half-lives, any comparable delays in all the releases would tend to be relatively more important in reducing the potential sumpwater and depressurization doses than the melt debris ones.

6.2.1.3 Time of Groundwater Contact

For sites with short groundwater travel times, the total population dose received via leaching of the melt debris would be somewhat dependent upon the time until substantial groundwater contact with the debris occurred. However, for sites with relatively long groundwater travel times, the total population dose would be essentially independent of the contact time for such times less than a few years.

6.2.1.4 Leach Rates

The leaching rates for the melt debris are among the more important but less well-known parameters in this study. Obviously, if the leaching rates of most of the potentially significant radionuclides were slow enough, then the consequences due to releases from the melt debris could be reduced to negligible levels.

The net effect of variation of the leach rates on the population dose is depicted in Figure 6.2 for a large lake site. The indicated leach rates in that figure are those for the fastest leaching elements, with the relative rates for all the other elements being as given in Table A12 in Appendix A. As can be seen in Figure 6.2, the population dose for releases from the melt debris is relatively insensitive to a rather wide range of leaching rates. The strong mutual dependence of the population dose on both the leaching rates and the groundwater travel time is not surprising inasmuch as an important factor in determining the resultant population dose is the amount of radioactivity which reaches wells or surface waterbodies; both these factors are significant in determining that amount.

The results shown in Figure 6.2 for the large lake site are representative of those found for sites at other bodies of surface water. That is, the plot of population dose for any other surface waterbody site as a function of leaching rate and groundwater travel time closely resembles that of Figure 6.2, with the most significant difference being the exact magnitudes of the doses received.*

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*There is also an observable difference in the groundwater travel times at which the curves such as those in Figure 6.2 start to descend rapidly for saltwater and freshwater sites because of the dominance of the population doses at those sites by different radionuclides (see Subsection 6.2.4).

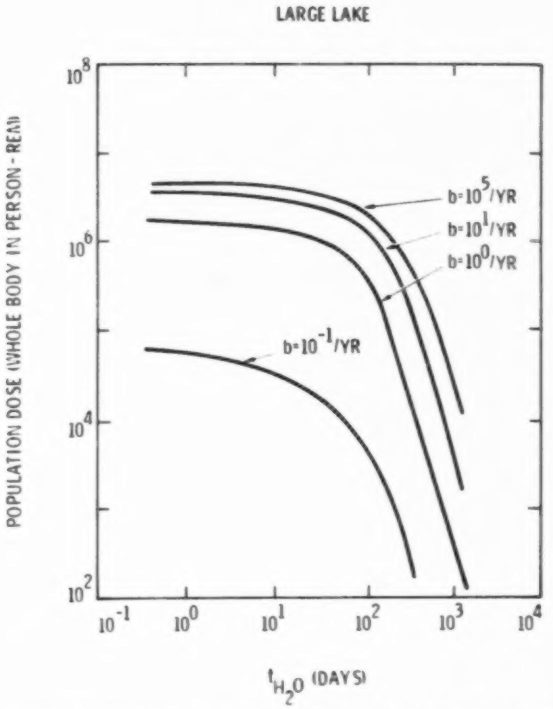

Figure 6.2 Population Dose from the Melt Debris after a PWR-7 Type Accident as a Function of Both Leach Rate and Groundwater Travel Times, with the Indicated Leach Rate Denoting that for the Most Rapidly Leached Element.

6.2.2 Population Dose as a Function of Radionuclide Travel Time

At a given site, the time it takes for a radionuclide to move from the vicinity of the accident to the accessible regions of the environment can be described in a first approximation in terms of three parameters: the average groundwater travel time from the reactor site to the nearest surface waterbody; the distribution coefficient of the element; and the dispersivity of the site. The first two parameters determine the average radionuclide travel time to the nearest waterbody; the last parameter indicates the spread of the radionuclide mass about that average. The shorter the average radionuclide travel time is, the more probable it is that significant human exposure can occur via the liquid pathways. Likewise, the larger the dispersion is, the greater the potential radiation hazard is.

Groundwater travel times for the reactor sites vary over many orders of magnitude, from times on the order of days to times on the order of many thousands of years or more. The appropriate value for the distribution coefficient for a specific element also varies significantly due to diverse local conditions; the variation among the values for different elements is likewise substantial. The dispersivities for different sites can be expected to vary over several orders of magnitude; even the variation at a given site can be significant.

6.2.2.1 Average Radionuclide Travel Time

The radiation dose tends to be a strong function of any factor which substantially alters either the average travel time for the groundwater or else the distribution coefficients of the dominant radionuclides. This dependence is displayed in Figure 6.3 while the fundamental reason for this relationship is shown in Figure 6.4.

As the latter figure indicates, the dependence of the estimated dose on the radionuclide travel time is due to variations in the amounts of radioactivity entering the open regions of the hydrosphere prior to decay. The fluxes of the radionuclides reaching the closest surface waterbody are, in turn, strongly related to the groundwater distribution coefficients. The two sides of Figure 6.4 differ by an order of magnitude in the values assumed for all the distribution coefficients. As can be seen, the corresponding net retardations of the indicated fluxes likewise differ by approximately an order of magnitude for all those nuclides with non-zero distribution coefficients.[*]

The calculated doses for a wide range of relatively short groundwater travel times (1 day to 10^2 days) are essentially equivalent for each of the generic surface water sites considered (for example, see Figure 6.2).[**] (This is because approximately the same amounts of the dominant radionuclides reach the surface water before substantial decay occurs for such groundwater travel times.[†]) As the groundwater travel times increase to 10^3 days, the resultant population doses for each of the sites decrease by approximately an order of magnitude.[**] (This decrease in the doses results from the significantly lower amounts of some of the dominant radionuclides reaching the surface water for these larger travel times.[†]) As the travel times increase beyond 10^3 days, the estimated population doses fall off extremely rapidly. At sites with such long travel times, the resultant consequences are compartively low. (This is due to the relatively small amounts of the dominant radionuclides reaching the surface water for such large travel times.)

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[*] It should be noted that for times after the accident which are larger than 100 years, the fluxes of the indicated elements passing across the interface between the groundwater and the surface water are negligible on the scale considered in Figure 6.4.

[**] For details, see Tables G10-G14 in Appendix G.

[†] See Figure 6.4

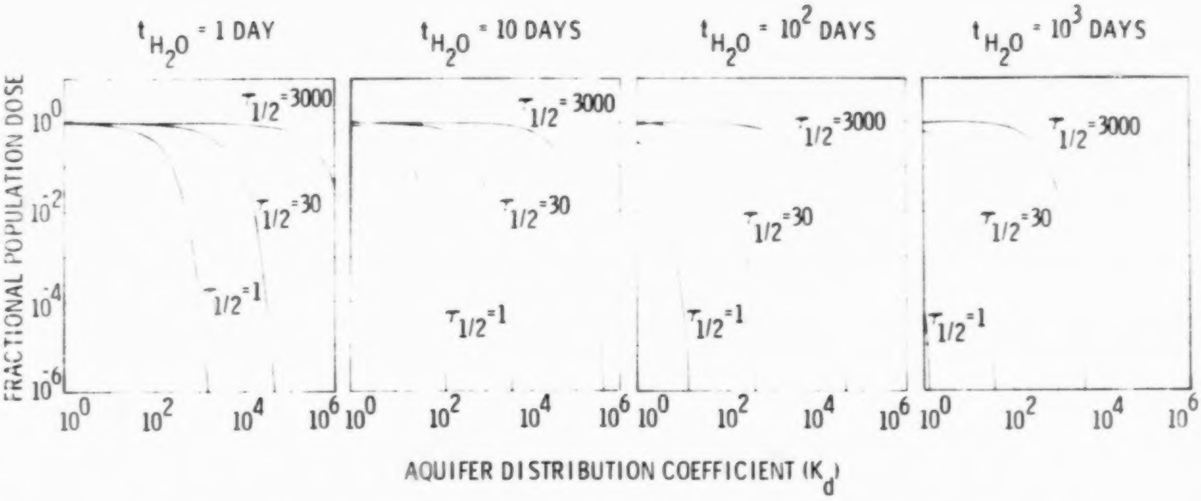

Figure 6.3 Relative Population Dose Received Via a Surface Waterbody as a Function of Groundwater Travel Time, Distribution Coefficient, and Radioactive Half-Life.

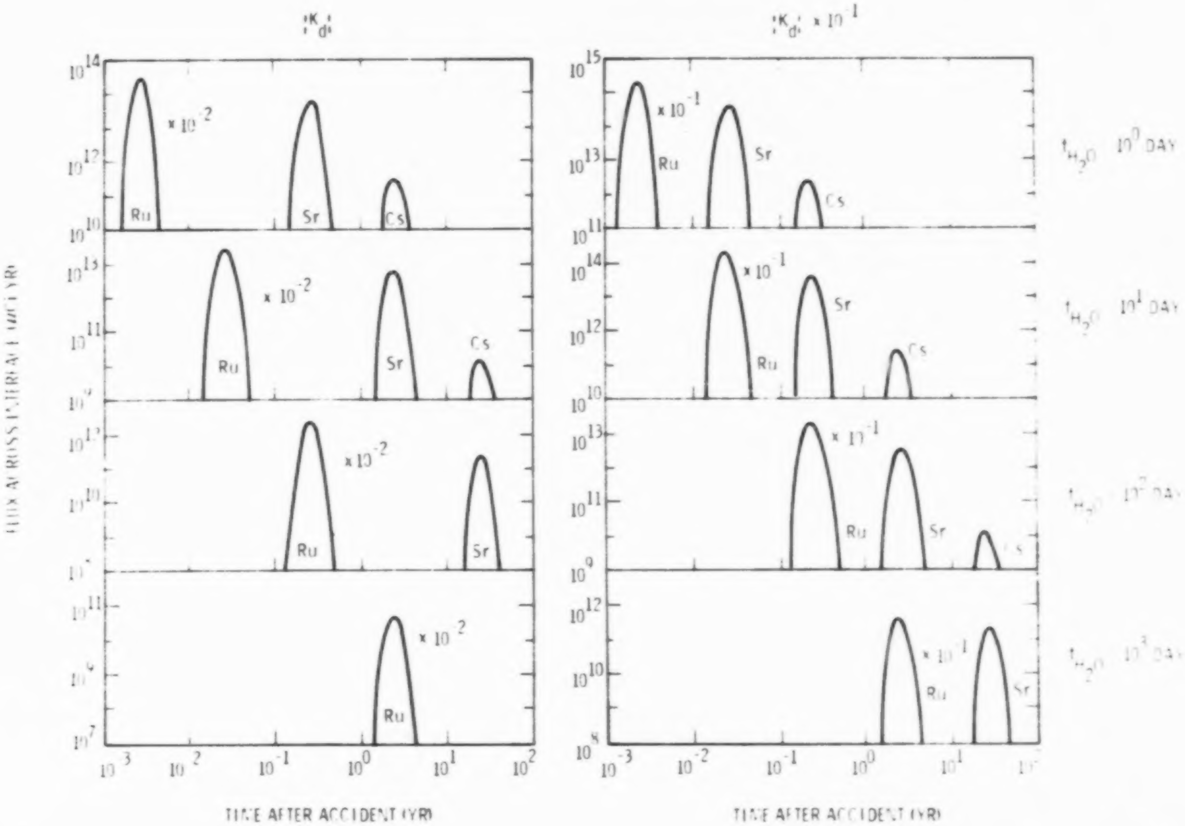

Figure 6.4 Fluxes of Radioactivity Passing Across the Interface Between the Groundwater and the Surface Water for Three Potentially Important Isotopes (Ru-106, K_d = 0; Sr-90, K_d = 20; Cs- 137, K_d = 200) as a Function of Groundwater Travel Time and Distribution Coefficient. (Instantaneous leaching of the melt debris of a PWR-7 type accident is assumed.)

Approximately half of the reactors are conservatively estimated to be located at sites which have groundwater travel times no less than 10^3 days, while about two-thirds are situated at sites which have times no less than 10^2 days. (See Chapter 3 and Appendix B.)

6.2.2.2 Dispersion

The dependence of the fluxes upon the groundwater dispersion coefficients is indicated in Figure 6.5 for average and low dispersion coefficients. The possibilities of more dramatic effects of dispersion variations are discussed in Section B4 of Appendix B.

6.2.3 Population Dose as a Function of Generic Site

For any given set of groundwater transport conditions, the total population doses for all the surface waterbodies except the ocean are approximately equivalent if the waterbodies are considered in isolation (see Table 6.2). The consequences at an ocean site are approximately one order of magnitude lower than those for reactors at other sites with equivalent groundwater transport conditions.

If the effects of other waterbodies farther downstream are taken into account, then the consequences can be ordered roughly as follows: river system, which flows into a significant estuary, which in turn, empties into the ocean; estuary, which empties into the ocean; large lake, which drains into another lake, and so forth;* and ocean. The details for waterbodies in series are considered in Appendices B and G.

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*The generic large lake was assumed to be Lake Michigan so that its flow was taken to be into Lake Huron, Lake Erie and so on.

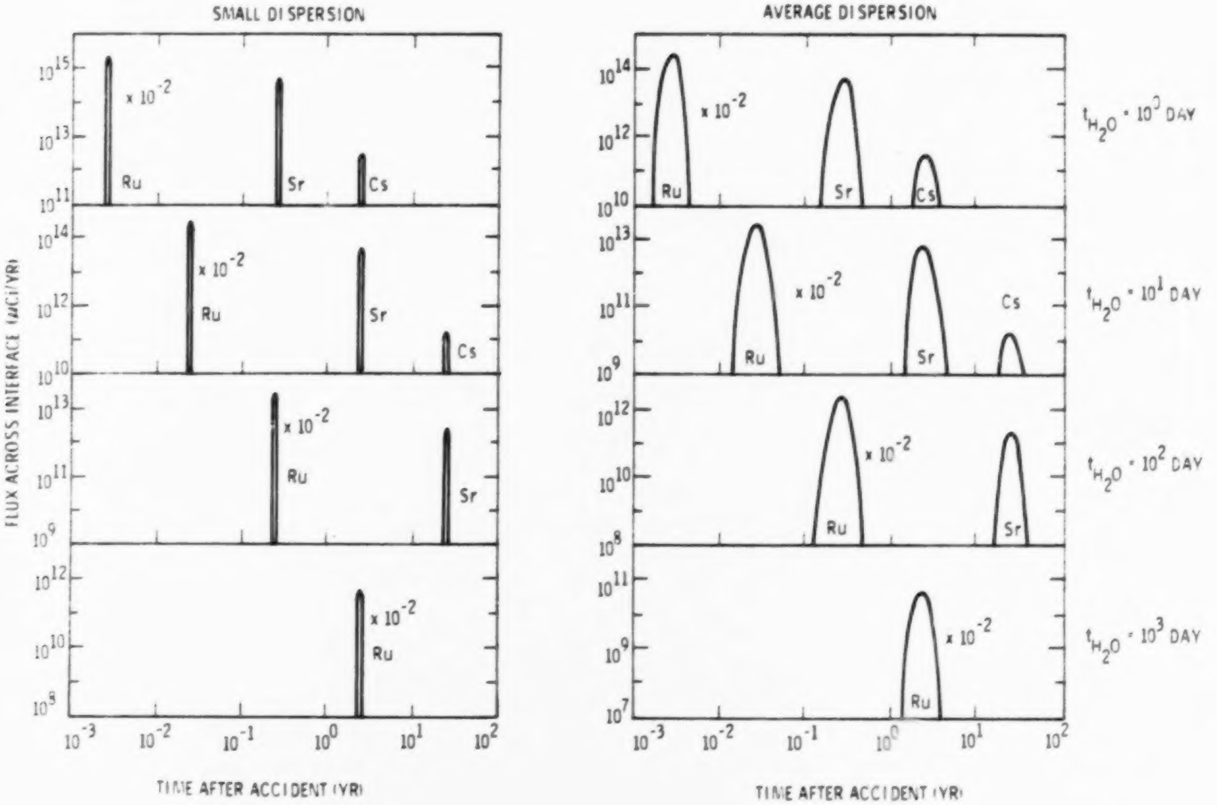

Figure 6.5 Fluxes of Radioactivity Passing Across the Interface Between the Groundwater and the Surface Water for Three Potentially Important Isotopes (Ru-106, Sr-90, and Cs-137) as a Function of Groundwater Travel Time and Dispersion Coefficient. (Instantaneous leaching of the melt debris of a PWR-7 type accident is assumed.)

Table 6.2 Estimated Population Doses for the Isolated Generic Sites(a,b,c)

Waterbody	Estimated Population Dose (Whole Body in Person-Rem)(d) Melt Debris	Sumpwater	Total
Large Lake	$4x10^{4}$ to $6x10^{6}$	$1x10^{3}$to $1x10^{7}$	$4x10^{4}$ to $2x10^{7}$
Nearshore	$2x10^{4}$ to $7x10^{5}$	$1x10^{3}$ to $2x10^{6}$	$2x10^{4}$ to $2x10^{6}$
Small Estuary	$2x10^{6}$ to $3x10^{7}$	$1x10^{5}$ to $4x10^{7}$	$2x10^{6}$ to $7x10^{7}$
Large Estuary	$1x10^{6}$ to $2x10^{7}$	$2x10^{4}$ to $2x10^{7}$	$1x10^{6}$ to $3x10^{7}$
Ocean	$1x10^{5}$ to $2x10^{6}$	$1x10^{4}$ to $1x10^{6}$	$1x10^{5}$ to $4x10^{6}$
Nearshore	$1x10^{5}$ to $2x10^{6}$	$1x10^{4}$ to $1x10^{6}$	$1x10^{5}$ to $4x10^{6}$
Free-flowing River	$3x10^{5}$ to $1x10^{7}$	$1x10^{4}$ to $5x10^{7}$	$3x10^{5}$ to $6x10^{7}$
Dammed River	$1x10^{6}$ to $3x10^{7}$	$1x10^{5}$ to $5x10^{7}$	$1x10^{6}$ to $8x10^{7}$

[a] The releases considered are for a PWR-7 accident and are assumed to occur instantly into the groundwater.

[b] For detailed results for isolated waterbodies, see Tables G10-G13 in Appendix G.

[c] For population doses for generic waterbodies in series, see Table G14 in Appendix G.

[d] Upper ends of the indicated ranges represent groundwater travel times of approximately 100 days or less and the lower ends represent times of 1000 days. Doses for larger travel times are much smaller.

6.2.4 Population Dose as a Function of Pathway

The relative importance of each of the various major exposure pathways to the population doses at each of the generic sites is indicated in Table 6.3. As can be seen in that table, the drinking water pathway is the largest contributor to the total population dose at all freshwater sites, whereas the aquatic food pathway is the major contributor to the population dose at all saltwater sites.

The most important elements contributing to the population doses are listed in Table 6.4 as functions of both the surface water type (salt or fresh) and the exposure pathway. As is indicated in that table, the dominant elements at saltwater and freshwater sites differ substantially for melt debris releases but not for sumpwater releases.

In addition to the total pathway doses and their compositions, the time-dependent buildups of these doses in the pathways are also important. The population dose at each of the generic surface water sites is depicted in Figure 6.6 as a function of both the time after the accident and the dominant exposure pathways for a single groundwater travel time. As can be seen in that figure, the time-dependent behavior of the dose in each pathway is extremely dependent upon the flushing characteristics of the waterbody. For example, the population dose for each of the pathways at a large lake site increases slowly as a function of time; in contrast, each of the pathway doses at a river system site increases relatively sharply over comparatively short periods of time.

Table 6.3 Relative Importance of Exposure Pathways at Each of the Generic Sites

Importance with Respect to Population Dose[a]

Waterbody	Drinking Water	Aquatic Food	Shoreline Usage
Isolated Aquifer	1	-	-
Large Lake	1	2	3
Estuary	-	1	2
Ocean	-	1	2
River System	1	2	3

Importance with Respect to Average Individual Dose(b)

Waterbody	Drinking Water	Aquatic Food	Shoreline Usage
Isolated Aquifer	1	-	-
Large Lake	2	1	3
Estuary	-	2	1
Ocean	-	1	2
River System	2	1	3

[a] Numerical estimates of the population doses for each important pathway at each considered waterbody are given in Tables G1-G8 and G10-G13 of Appendix G.

[b] Numerical estimates of the average individual doses for each important pathway at each considered waterbody are given in Tables G15 and G16 of Appendix G.

Table 6.4 Most Important Elements as Functions of Type of Water and Exposure Pathway(a)

	Fresh Water		Salt Water	
	Melt Debris	Sumpwater	Melt Debris	Sumpwater
Drinking Water	Sr	Cs	-	-
Aquatic Food	Sr, Cs	Cs	Ru, Co	Cs
Irrigated Food	Sr	(Cs)	-	-
Shoreline	Ru, Cs, Co	Cs	Ru, Cs, Co	Cs

[a] It should be noted that the most important elements also tend to be functions of the groundwater travel time. The indicated elements are for a groundwater travel time of 10 days. For travel times of 100 days and more, Cs and Co contribute insignificantly to the total population dose. For travel times of 1000 days and more, Sr also adds little to that total dose. For travel times less than 10 days, elements other than those indicated above can be significant contributions to the population dose.

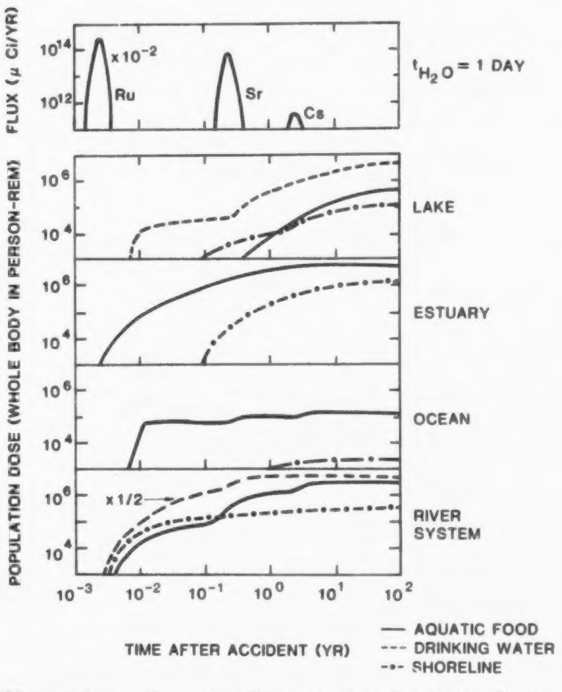

Figure 6.6 Time-Dependent Behavior of the Population Dose as a Function of Generic Site Type and Exposure Pathway for a 1-Day Groundwater Travel Time. (Instantaneous leaching of the melt debris of a PWR-7 accident is assumed.)

6.2.5 Population Dose as a Function of Site-to-Site Differences

The population doses estimated to be received at actual sites could depart from those obtained for the generic sites due to at least the following differences: sizes, designs, and statuses of reactors; applicable release rates, including leaching rates; subsurface hydrologies, especially, permeabilities, effective porosities, hydraulic gradients, distances to the nearest surface waterbodies, and soil types; surface hydrologies, in particular, flushing rates, dilution characteristics, sedimentation rates, and sediment properties; relevant aquatic ecosystems; and sizes and compositions of affected populations for the various pathways. The reasonable alterations in some of these factors could easily result in differences of orders of magnitude in estimated doses. The results reviewed in the earlier chapters and in the previous subsections of this chapter give indications of the effect of any variations of these factors upon the estimated population doses. More extensive sensitivity analyses of the parameters utilized for release rates, subsurface movements and surface water considerations are presented in Appendix B.

6.3 Individual Dose

The previously-considered population dose can be used as one indicator of the consequences. However, it is also interesting to consider the corresponding individual doses.*

6.3.1 Individual Dose as a Function of Generic Site

Although the population doses for all the generic surface waterbodies other than the ocean are essentially the same, the doses expected to be received by average individuals at the various sites differ substantially. If the doses received by average individuals resulting from a meltdown accident at a site with a 1-day groundwater travel time are considered, estimates of the maximum* doses received by such individuals can be obtained. As can be seen in Table 6.5, the maximum* values of both the "average" individual doses and the corresponding "average" dose rates tend to be largest for waterbodies, or regions of those waterbodies, with relatively high concentrations of radionuclides.

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*Because the populations-at-risk for the liquid pathways are very difficult to describe in detail, it is not easy to consider doses to individuals. Furthermore, many averaging techniques have been employed in the modeling performed in this study so that pathway concentrations are not really obtained in sufficient detail to permit consideration of actual individual doses. However, it is still interesting to consider the typical dose received by some fictitious "average" individual.

The estimated individual doses and dose rates in Table 6.5 are for relatively rapid releases of material into the hydrosphere. For somewhat slower releases of activity, the individual doses are usually not significantly changed but the corresponding dose rates can be substantially lowered.

6.3.2 Individual Dose as a Function of Pathway

The dependence of the average individual doses on the type of pathway is indicated by the values presented in Table 6.5. The individual doses are significant only for ingestion of drinking water, ingestion of aquatic food and shoreline usage.

As can be seen in Table 6.3, the pathway at a given site with the largest average individual dose is not necessarily the same as the pathway at that site with the largest population dose. Whereas the estimated population doses in this study are dominated by the drinking water pathway at all freshwater sites and by the aquatic food pathway at all the saltwater sites, the estimated average individual doses are dominated by the aquatic food pathway at all the surface water sites considered.

6.4 Health Effects

The estimation of the resultant health effects is extremely dependent upon both the characteristics of the affected waterbody and the properties of the utilized health-effects model. This is due to the dependence of the estimated effects on the rates at which the individual doses are received. (See Appendix D.) Because of the difficulties in describing the populations-at-risk for some of the liquid exposure pathways, most of the health effects have been calculated using the linear model.

For the ingestion pathways, the organ-by-organ estimate in the linear model results in approximately 250 fatalities per 10^6 person-rem of dose.*·** In contrast, the estimation of ingestion-related fatalities by whole-body doses predicts 122 fatalities per 10^6 person-rem.[2] The organ-by-organ estimation of the fatalities due to external exposure is the same as that for whole-body considerations.[2]

The maximum† numbers of cancer fatalities estimated for releases from the melt debris alone are presented in Table 6.6.†† The estimated numbers of fatalities at many sites would be considerably less than these values.‡

6.5 Comparison to the Risk of Releases to the Atmosphere

One possible comparison of the risk from releases to the hydrosphere to that from releases to the atmosphere after a meltdown accident can be made by considering the complementary cumulative distribution functions‡‡ (CCDF's) of the expected latent cancer fatalities for the two types of releases. As can

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*Maximum denotes largest for the groundwater travel times explicitly considered; larger values would be obtained if groundwater travel times less than 1 day were utilized.

*When ruthenium is a significant contributor to the dose, then the number of fatalities as estimated on an organ-by-organ basis can exceed this by more than a factor of two. (See Table 6.6.)

**In general, the largest internal doses from releases to the hydrosphere are estimated to be received by the skeletal tissues, the bone marrows, and the lower large intestines (see Tables G1-G8). Therefore, the cancers associated with exposures of these parts of the body (bone cancer, leukemia, and GI tract cancer, respectively) tend to be present in larger numbers when an organ-by-organ estimate is used than might otherwise be expected (see Table G9).

†See the footnote in Subsection 6.3.1 for the meaning of maximum as used here.

††It should be noted that even the maximum numbers represent only small increases of the cancer rates above the normally-expected background rates.

‡It should be mentioned that the cancer estimates given in this report and in the RSS are for fatalities only;[2] that is, the estimated total number of cases of cancer would always be significantly larger than the numbers given in Table 6.6.

‡‡CCDF's plot the probability of exceeding a given number of consequences versus the number of consequences.[1]

Table 6.5 "Maximum" Average Individual Doses and Dose Rates as a Function of Generic Site[a,b,c,d]

	"Maximum" Average Dose (rem) Pathway								"Maximum" Average Dose Rate[e] (rem/yr)	
	Drinking Water		Aquatic Food		Shoreline		Total			
Waterbody	Debris	Total	Debris	Total	Debris	Total	Debris	Total	Debris	Total
Large Lake										
Nearshore	4	8	1	14	<1	<1	5	22	5	22
Rest of Lake[f]	<1	1	1	7	<1	<1	1	8	<1	<1
Estuary										
Small	-	-	5	7	3	14	8	21	8	21
Large	-	-	1	2	<1	3	2	4	2	4
Ocean Coast										
Nearshore[f]	-	-	1	1	<1	<1	1	1	1	1
Offshore	-	-	<1	<1	-	-	<1	<1	<1	<1
River System										
Free-Flowing	2	5	4	60	<1	<1	7	65	7	65
Dammed	5	8	6	70	<1	1	12	78	12	78

[a] A groundwater travel time of 1 day is assumed. Both leach and sumpwater releases (for a PWR-7 accident) are taken to occur essentially instantly into the ground.

[b] The corresponding doses for groundwater travel times other than 1 day are presented in Tables G15 and G16 of Appendix G. In general, the values for the "average" individual doses as functions of the groundwater travel times follow curves like those presented in Figure 6.1 for the values for the total population doses. As expected, the relative scales are the same for both the individual and the population doses at a given type of site.

[c] The populations exposed to the different exposure pathways at a given site do not necessarily consist of the same individuals. Furthermore, the doses delivered by exposure to various radionuclides can occur over much different periods of time. Thus, this consideration of doses to individuals is rather artificial. The values given in this table have been obtained using three assumptions: (1) that the individual is average with repect to usage habits for all the liquid pathways; (2) that all the individual's usage for these pathways involves only the contaminated waterbody; and (3) that the individual lives long enough to be exposed to all the dominant radionuclides. It should be noted that individuals are much more likely to have approximately average usage habits for the dominant pathways at freshwater sites than for those at saltwater sites.

[d] For all waterbodies except river systems, the maximum values of the "average" doses are approximately equal to the average values of the "average" doses. In contrast, the actual doses received by the "maximum" individuals would be considerably higher than those indicated for the maximum values of the "average" individuals for all cases due to two factors: (1) the maximum individual is defined to have higher usage rates for all the exposure pathways than the average individual; and (2) the maximum individual would usually experience higher concentration of radioactivity in each of those pathways than the average individual would.

[e] Assumes that the dose is delivered over at least a one-year period; the actual dose rates for very rapid releases into fast flushing waterbodies could even be higher than the indicated rates.

[f] Most of the population dose for this waterbody comes from this region.

Table 6.6 Maximum Number of Total Cancer Fatalities as a Function of Both Generic Site and Pathway[a,b,c]

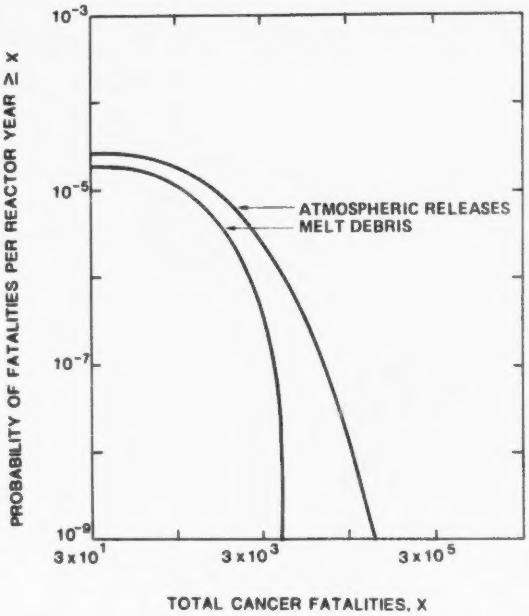
	Total Fatalities From Melt Debris Releases[b]					
	Pathway					
	Drinking Water		Aquatic Food		Shoreline	
Waterbody	**W. Body**	**Sum**	**W. Body**	**Sum**	**W. Body**	**Sum**
Large Lake						
Nearshore	49	150	10	17	16	16
Rest of Lake	540	1300	49	84	39	39
Estuary						
Small	--	--	1600	6700	1300	1300
Large	--	--	1600	6200	320	320
Ocean Coast						
Nearshore	--	--	260	1300	7	7
Offshore	--	--	35	110	--	--
River System						
Free-Flowing	1100	3800	370	650	57	57
Dammed	2200	8700	500	1000	100	100

[a] A groundwater travel time of 1 day is assumed. Leaching of the melt debris (for a PWR-7 accident) is taken to occur instantly. The effects of sumpwater releases are not included.

[b] The linear model for health effects is used. Total fatalities are calculated in two ways: using whole body doses and summing over the individual organ doses.

[c] The corresponding total health effects for groundwater travel times other than one day are presented in Table G9 of Appendix G. The health effects for the various pathways for the different groundwater travel times can be estimated from the information given in Tables G1-G8 and the health effects tables presented in Appendix D. In general, the values of the total number of cancers as calculated from the whole-body doses and as taken as functions of the groundwater travel times follow curves like those presented in Figure 6.1. The health effects for combined debris and sumpwater releases would generally be larger than those estimated for the effects of debris alone and can be estimated either by combining the information in Tables G10-G13 with that in the health effects tables in Appendix D, or by comparing the corresponding values for debris only and for debris and sumpwater combined are given in Tables G15 and G16, respectively, and using the comparison to extrapolate from the health effects estimates presented in Table G9.

be seen in Figure 6.7, the estimated CCDF for such fatalities for releases from the melt debris* lies below that for atmospheric releases.[1,3,**,†,††,‡] However, it should be noted that the straightforward comparison of the two CCDF's in Figure 6.7 is somewhat misleading because the curve presented in the RSS for atmospheric releases includes many credits for various interdictive procedures whereas that given here for hydrospheric releases does not include any such credits.‡‡ Therefore, the estimated number of fatalities predicted at any given probability for the atmospheric releases in the RSS will appear to be substantially smaller than that estimated here for hydrospheric releases for situations of actual comparable risk. In any case, it can be seen that the potential risk to the human population from releases to the hydrosphere after a meltdown is non-negligible relative to that from releases to the atmosphere.



ATMOSPHERIC RELEASE:: for atmospheric releases if the results presented in the Reactor Safety Study are assumed;

MELT DEBRIS: for debris releases if it is assumed that no interdictive procedures are instituted and that for one-tenth of the river systems the consequences are as estimated in this report for a heavily-dammed system connected to a significant estuary and for the rest of the rivers the consequences are an order of magnitude smaller.

Figure 6.7 Complementary Cumulative Distribution Functions for Latent Cancer Fatalities After a Meltdown Accident.

6.6 Conservatisms

In this study, an attempt has been made to consider in as realistic a manner as is possible the potential consequences resulting from the releases of radioactivity to the hydrosphere after a meltdown accident. Where realistic values for important parameters were not available with an effort in line with the required limited scale of this project, the consequences have usually been investigated as functions of reasonable ranges of these parameters (see Table 6.7). Throughout this study, conservative values of most questionable sensitive parameters have been employed for the basic calculations. Thus, in one sense, the estimates of the consequences presented here can be considered to be upper bounds of the actual values.[*],[**]

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*Only the releases from the melt debris are included here because of the large uncertainties associated with both the probabilities and the magnitudes of sumpwater releases (see Appendix A).

**The CCDF for atmospheric releases was produced by utilizing the central estimate for latent cancers and a summation over individual organ doses whereas the CCDF for hydrospheric releases was generated using the upper estimate for such cancers and a consideration of only whole-body doses. However, because the effect of using the central estimate relative to using the upper estimate is a decreasing of the calculated atmospheric fatalities by a factor of approximately two,[1] and because the effect of using a sum over individual organ doses relative to using a whole-body dose is an increasing of the calculated atmospheric fatalities by a factor of about two,[1] the different procedures employed for estimating atmospheric and hydrospheric fatalities in the two studies are in some senses equivalent.

†The magnitudes of the consequences in the atmospheric CCDF given in the RSS are slightly too small, especially at the low-probability, high-consequence end of the curve due to calculational problems.[3]

††It should be noted that this is the situation only if the river systems described in this study are taken to be worst cases. If the river systems used in this study are taken to be generic, then the CCDF for releases to the hydrosphere lies above that given for atmospheric releases for small and moderate consequence magnitudes.

‡The uncertainty band for the hydrospheric CCDF is wide. However, its exact width is currently not quantifiable.

‡‡It is thought that the credits taken in the RSS may easily lower the estimated number of fatalities at a given probability by as much as an order of magnitude or more.

In contrast to the conservative values of most parameters, the values of some of the other parameters employed in the modeling in this study do not necessarily cause the resultant radiation doses to be overestimated. In particular, the use of generic surface waterbodies and corresponding generic populations-at-risk may not be conservative.[†]

Another possible non-conservative feature of the modeling in this study is the use of pathway models which do not necessarily err on the side of overestimating the resultant radiation doses.[††] In addition, the use of approximated data to estimate the groundwater travel times for some of the sites did not necessarily result in times which were relatively low.[‡] The effects of all three of these possibly non-conservative aspects of the modeling in this study are difficult to quantify.

6.7 Limitations

A basic problem of any generic study is that representative behavior must be identified throughout the system being considered. In particular, all the critical portions of such a study must be chosen so that they represent typical systems. However, the definition of representative or typical is not always obvious. Furthermore, the appropriate distributions of the parameters required to describe the system are not always known; as a result, even if what is representative can be adequately defined, it often cannot be determined. In addition, there may not even "exist" an actually representative system.[*]

This study has attempted to obtain distributions of the values of some of the critical parameters for the liquid pathways. In particular, a zeroth-order attempt has been made to estimate the appropriate distributions of some of the relevant groundwater transport parameters. However, it must be acknowledged that the uncertainties in the resulting distributions are large. The determination of other significant, but less sensitive, factors has not been explored in as much depth due to the limited scope of the present study. (See Subsection 1.2.3 in Chapter 1.)

Another type of limitation of this study is related to shortcomings of the various models employed to perform the calculations. The limitations of the specific models are discussed in the various appendices which describe them.

6.8 Summary

In contrast to the situation for releases to the atmosphere, the most probable RSS meltdown categories result in the largest releases to the hydrosphere. Furthermore, significant amounts of radioactivity are generally expected to be released to the hydrosphere during almost any meltdown accident involving complete melt-through of the containment basemat.

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[*] Among the most sensitive sets of parameters (see Table 6.7) for which conservative estimates have been used are the following: (1) the rates of the releases to the hydrosphere; (2) the groundwater distribution coefficients, especially that for ruthenium; and (3) the numbers of sites with low groundwater travel times. Lower values of the first set of these parameters and/or higher values of the second could significantly decrease the magnitude of the estimated consequences at any site. Lower values for the third of these parameters could substantially decrease the number of sites at which the consequences for a given accident are estimated to be potentially significant.

[**] As was noted in Chapter 2, it is possible that the melt might not completely penetrate the containment basemat as a result of a meltdown accident. In such a case, there would not usually be any releases of radioactivity directly to the hydrosphere. (See Subsection A4.3 of Appendix A.) Therefore, because this study has assumed that complete melt-through would always accompany any meltdown accident, the results presented here may be conservative.

[†] Inasmuch as the actual distributions of the relevant parameters for describing these waterbodies were generally not obtained, the values which would be representative are not known. The problem of the representativeness of the generic waterbodies is probably most in question for the consideration of river systems. For these waterbodies, there exists a very wide variety of possible descriptions and related parameters. Unfortunately, more reactors are found near river systems than are located near any other type of surface waterbody.

[††] For example, see Appendix C and the related report for discussions of the possibly non-conservative aspects of the aquatic food models and see Appendix D for similar considerations of the shoreline model.

[‡] See Section B2 in Appendix B

[*] The consideration of only "representative" systems sometimes does not permit adequate inclusion of the behavior of nonrepresentative systems. In such cases, the average of the behavior of the actual systems may not correspond very closely to the behavior of the best generic model.

Table 6.7 Sensitivity of Population Doses to Basic Parameters

Parameter	Sensitivity	Necessary Range of Consideration	Availability of Data
Magnitudes of Releases	Very sensitive but straightforward	-	Magnitudes of initial releases available for a few reactors(a)
Rates of Releases	Sensitive if slow enough that significant decay occurs during release(b)	A range of release rates must be considered	Only generous bounds for rates available(a)
Groundwater Travel Times	Very sensitive(b)	A range of radionuclide travel times must be considered	Often not available (the distribution of groundwater travel times has been estimated)
Groundwater Distribution Coefficients	Very sensitive(b)	See above comment	Not generally available for a wide range of radionuclides at most sites
Surface Water Dilution and Flushing Rates	Sensitive(b)	A range of rates must be considered; for some types of surface waterbodies, generic values can be estimated	Generally available (the distribution of necessary surface water parameters has not been estimated for most types of sites)
Sediment Distribution Coefficients	Sensitive only if large enough(b,c)	Effects caused by "large enough" values should be considered	Not generally available for a wide range of radionuclides at most sites
Other Sediment-Related Parameters	Sensitive only if large enough(b,c)	Effects caused by "large enough" values should be considered	Often adequately available for simple models but seldom available for complex ones
Concentration Factors, Processing Factors,...	Very sensitive but straightforward(d)	-	Not generally available for a wide range of radionuclides at most sites
Sizes of Populations-at-Risk	Very sensitive but straightforward	-	Population usage rates generally available; in contrast, the usage rates with respect to individuals not available for some pathways

[a] Many uncertainties remain in the description of the releases to the hydrosphere. Included in the poorly characterized parts of the description are the following: all the factors relating to basemat melt-through; the magnitudes of the releases; the structure of the interface around the melt debris; the rate of cooling of the debris; the availability of groundwater for leaching; the character of the debris; and the availability of alternate escape routes of the sumpwater from the containment.

[b] See Section B4 in Appendix B for details.

[c] Large enough implies that the concentrations of the radionuclides are significantly lowered as a result of sediment-nuclide interactions.

[d] Assumes the use of simple pathway models; for more complicated models, the results are very sensitive but not necessarily straightforward (see Appendix C and the related report).

At approximately 35% of all the reactors considered, the radiation dose to the entire human population as a result of releases to the hydrosphere via the hole in the containment basemat formed by the melt is estimated to be very small (in general, much less than 10^4 to 10^5 person-rem). At another 25% of the reactors, the resultant radiation dose is estimated to be no more than 1 to 9×10^6 person-rem, while at the remaining 40% the potential dose is no larger than 1 to 10×10^7 person-rem for such releases.*

The corresponding dose to an average individual over an entire lifetime as a result of the releases to the hydrosphere (via the melt hole) is estimated to be less than 1 rem at approximately 60% of all the reactors under discussion. Furthermore, the total dose to such an individual is estimated to be no more than 1 to 10 rem at another 20% of the reactors and no larger than 100 rem at the remaining 20% of the reactors.*

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*The primary contributor to this category (sites containing 65 out of 73 reactors) consists of river system sites. As has been previously noted, these sites are probably least well described in a generic sense.

*See footnote for previous paragraph

These percentages for both population and average individual doses are generally thought to be conservative. However, they do not take into account the effect of various design features which might permit the early emergence of radioactivity into the hydrosphere at certain sites and, hence, result in a significant dose at any site possessing such features. Furthermore, they do not include the potentially significant effects of direct entry of sumpwater into surface water.

Variations of conditions at actual sites could easily result in the doses being either higher or lower than those obtained in this (generic) study by at least an order of magnitude. Likewise, reasonable variations in certain portions of the modeling could result in estimated doses which differed from those presented here by an order of magnitude. Thus, the estimated doses and their related percentages-of-occurrence have rather large uncertanties associated with them.

The calculations indicate that if interdictive (mitigating) actions are not taken, then the liquid pathways can perhaps contribute significantly to the risk related to a core meltdown accident.

References

[1] U.S. Nuclear Regulatory Commission, "Reactor Safety Study," WASH-1400, NUREG-75/014, Main Report, USNRC, 1975.

[2] U.S. Nuclear Regulatory Commission, "Reactor Safety Study," WASH-1400, NUREG-75/014, Appendix VI, USNRC, 1975.

[3] I. B. Wall, S. S. Yaniv, R. M. Blond, P. E. McGrath, H. W. Church, and J. R. Wayland, "Overview of the Reactor Safety Study Consequence Model," *Nuclear Systems Reliability Engineering and Risk Assessment*, J. B. Fussell and G. R. Burdick, Editors, Society for Industrial and Applied Mathematics, Philadelphia, 1977.

7 Effects of Mitigating Actions

7.1 Introduction

In theory, interdiction can be employed either close to the accident site so that exposure of the entire environment is minimized, or else selectively farther away so that the exposure of just certain portions of the human population is lessened for specific exposure pathways. Actions of the former type are called source interdictive procedures while actions of the latter type are called pathway interdictive procedures.

Discussions of the various methods which might be employed to mitigate the radiation dose to the human population, along with considerations of the feasibilities and costs of these methods, are presented in Appendix F. This chapter merely reviews the salient features of that appendix.

7.2 Source Interdiction

7.2.1 Methods of Source Interdiction

Potential methods to contain radioactive material within the area surrounding a reactor following a core-melt accident include the following: injecting water to form a barrier to groundwater flow; withdrawing contaminated water for treatment; lowering the watertable to isolate the melt debris; and installing a grout curtain to contain the debris.

7.2.2 Feasibility of Source Interdiction

The possibility of successful source interdiction depends on several factors, including at least the type(s) of releases to the hydrosphere, the subsurface conditions at the site, and the presence of certain engineered features. Source interdiction is generally most feasible for the melt debris. The feasibility of sumpwater interdiction would depend on both the escape path taken by the water and the conditions under the containment. Obviously, the direct release of sumpwater to any surface water would preclude the interdiction of that release at the source. The rate of release to the subsoil, and hence the possibility of successful interdiction, for both sumpwater and depressurization releases would depend upon the permeability of the soil and upon the level of the watertable with respect to the level of the bottom of the containment floor. The presence of certain engineered features, such as underdrain systems, might obviate the possibility of sumpwater interdiction.

At their best, source interdiction methods, such as those considered in Appendix F, are long-time, but not necessarily permanent, solutions. For example, in the case of grouting, an initial attempt might not result in a continuous barrier. Furthermore, the grouting could fail eventually by deterioration of the grout itself or as a result of seismic activity. In time, the melt and its grout curtain might be resealed inside another grout barrier. Alternatively, the melt debris might be moved to a permanent waste repository.

7.2.3 Benefits and Costs of Source Interdiction

The potential benefits of successful source interdiction would be the elimination of all adverse consequences associated with the releases to the hydrosphere. Against the benefits would need to be balanced the costs: possible doses to the workers implementing the interdictive actions; potential accidental early releases of radioactivity to the environment, for example, by pumping of contaminated water from the ground to the environment; and the monetary costs of taking the actions. If significant consequences were expected, then the benefits would probably outweigh the costs. The costs might be justified even if only low levels of consequences were anticipated.

7.3 Pathway Interdiction

In situations with relatively rapid releases to accessible portions of the hydrosphere, source interdiction might not be possible. Furthermore, certain siting considerations might render such procedures infeasible. In these circumstances, the only mitigating procedures which would be possible would be pathway interdictive ones. Pathway interdictive procedures might also be required if source interdictive procedures either were not executed for some reason or else were not successful in containing the radioactivity.

The least costly type of pathway interdiction involves interruption of the flow of radioactivity at some point along the exposure pathway, usually close to the domain of contact with humans. In general, only the human population benefits from such actions; the rest of the population in the environment do not. Examples of such interruptive procedures are substitution of water supplies, confiscation of finfish and shellfish, and relocation of certain portions of the population.

More costly pathway interdictive procedures involve restoration of some or all of the environment to an "uncontaminated" state comparable to that before the accident. Ultimately, most or all populations tend to benefit from such actions. Examples of such restorative procedures are dredging of sediment and cleaning of shorelines and floodplains.

7.3.1 Methods of Pathway Interdiction

Possible pathway interdictive procedures include those just mentioned plus treatment of water, denial of water for some purpose (e.g., irrigation), denial of access to contaminated areas, and so forth. Some of the potentially more important methods which might be implemented along the exposure pathways are considered in more detail in Appendix F.

7.3.2 Feasibility of Pathway Interdiction

The possibility of successful pathway interdiction depends on several factors, including at least the type(s) of release(s) to the hydrosphere, the concentration of radioactivity in the water, the degree of localization of the activity, and the characteristics of the relevant exposure pathways. Pathway interdiction is generally most feasible for situations in which the radioactivity is most localized, and, therefore, most concentrated. This feature tends to limit the scale of the operations required. As a result, any necessary actions are both more tractable and more economical. Furthermore, it tends to result in higher radiation doses per capita so that any actions taken are potentially more effective in lowering the overall population dose.*,**

For analogous reasons, releases of shorter duration are generally more easily and economically interdicted. Furthermore, any actions taken after such releases tend to be more efficacious in lowering the total population dose because of the greater initial localization of the radioactivity. At least for some waterbodies, a release of short duration would not require repeated mitigating actions to be taken over an extended period of time; the resulting actions might need to be taken only once or else just a few times.

Pathway interdiction would be easier if fewer pathways needed to be interdicted. In general, the drinking water pathway would be the most straightforward major pathway to interdict.

7.3.3 Benefits and Costs of Pathway Interdiction

The potential benefits of successful pathway interdiction would be the reduction of some of the consequences associated with the releases to the hydrosphere. Although a complete elimination of all the consequences is theoretically possible by implementation of pathway interdictive procedures, such a reduction would be practically impossible at any surface waterbody.

Against the potential benefits would need to weighed the possible costs: disruption of people's lives, for example, by loss of livelihood or recreational facilities; disturbance of the ecosystem which might not be too adversely affected by the radiation itself; and possibly substantial monetary costs. Even if a significant number of adverse health effects were anticipated, the costs of pathway interdiction might be prohibitively high at many surface waterbodies. The costs would depend on the possible effectiveness of the procedures. This, in turn, depends upon the hydrological characteristics of the affected water body(ies).

7.4 Summary

Source interdictive procedures would always be preferable to pathways ones. Relative to pathway procedures source procedures could usually be described as being the following:

easier: the actions required at the radioactive source would usually be of a much smaller scope than most pathway procedures;

more effective: mitigating actions taken close to the source would have the potential of completely eliminating the radiation exposure of the human population whereas pathway procedures would never be that effective, except at "dry" sites;

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*In general, practical pathway interdiction criteria are based on the dose to an individual. Therefore, at sites with hydrologies which favor low concentrations, interdiction would not be as likely to be implemented.

**The current regulations and guidelines concerning limits for radiation doses resulting from accidental releases of radioactivity are reviewed in Addendum F3 in Appendix F. The estimated maximumn average doses resulting from a meltdown accident at any of the generic waterbodies considered in this study are given in Table 6.5 in Chapter 6. By comparing the limits and the estimated doses it can be seen that pathway interdictive procedures often would not be legally required for the liquid pathways, except perhaps in the vicinity of the site of the accident.

more permanent: source procedures would not need to be repeated as often as some pathway procedures would need to be; and,

less expensive: the actions required could be performed on a very localized scale instead of over a large part of the affected environment.

Successful source interdiction could be expected to virtually eliminate the radiation doses received by the human population at many sites. For at least 50% of the sites, and probably for a much larger percentage, there would be sufficient time to employ such procedures at the source. However, they would not necessarily be possible at all of the sites due to both subsurface characteristics and engineered features existing at some sites. Furthermore, they would not be possible for direct releases of sumpwater into the accessible portions of the environment.

It is possible that the lack of knowledge of the subsurface region at some sites might significantly hamper any source interdictive procedures implemented at those sites. For sites at which relatively rapid action would be necessary, such ignorance could conceivably preclude, or at least delay, successful source interdiction.

For situations in which source interdictive procedures were not either feasible or successful, then the radiation dose could be mitigated only by actions taken along the exposure pathways. The effectiveness of such procedures would depend upon the characteristics of the affected waterbody(ies). Only at a dry (isolated aquifer) site could such procedures be completely effective.

APPENDIX A

Core-Melt Source Terms

APPENDIX A

Core-Melt Source Terms

Authors: W. B. Murfin
S. J. Niemczyk

A1 Introduction

The radioactive content of the melt is determined essentially by the fission products, actinides and activated structural materials present in the reactor at the time melt-through is initiated. The overall physical and chemical characteristics of the final melt are determined primarily by the various nonradioactive materials taken into the melt by accretion.

In this appendix, the chemical makeup of the melt is considered initially as a function of the dominant reaction conditions present during meltdown; in particular, these conditions are related to the relevant accident sequence. After that, the ranges of melt compositions which can result from differences in the incorporation of reactor structural materials for various sequences are discussed; also, the effect of steam explosions upon the addition of structural materials to the melt is included. Next, the various melt makeups which result from contact with two extremes of concrete composition are given. Finally, the effects of the soil and groundwater underlying the reactor containment are added.

After the general composition of the final melt is considered, the partitioning of the radioactive materials into atmospheric and nonatmospheric portions is reviewed. In particular, the three types of releases into liquid pathways which result from deposition of radioactive material directly into the substratum at the accident site are considered.

In this appendix, the emphasis is on the quantitative aspects of the descriptions of the possible radioactive releases to the hydrosphere after a core-melt accident. Related qualitative aspects of that description are reviewed in Chapter 2.

A2 Radioactive Inventory of Reactor Core

A2.1 Method

The radioactive source (fission products, actinides and structural materials) in the reactor immediately prior to the initiation of the core melt was obtained using ORIGEN.[1] To facilitate more meaningful comparison between the liquid pathway consequences presented here and the atmospheric pathway consequences given in the Reactor Safety Study (RSS), the same ORIGEN-generated inventory that was used in the RSS has been employed in this study. That is, the changes in the ORIGEN-produced inventory due to updating of the ORIGEN code since the RSS was performed have not been considered. Furthermore, the changes in the radionuclide inventory which would be obtained by using more realistic fuel cycle models in conjunction with the ORIGEN code as compared to the model used in the RSS have not been addressed.

A2.2 Reactor Status

As was discussed in the Reactor Safety Study, the radionuclide inventories were calculated for a 3200-thermal megawatt (MW) three-region pressurized water reactor (PWR) of a typical four-loop Westinghouse design. The core was assumed to operate at an average power density of 40 MW/MTU. (MTU is defined as metric tons of uranium initially charged to the reactor.) Inventories were calculated for an equilibrium core initially charged with 3.3 percent enriched uranium with average burnups of the three regions of 8800, 17,600 and 26,400 megawatt-days (MWD)/MTU.

Inasmuch as PWR's typically operate at power densities of 30 to 37 MW/MTU, the radionuclide inventories used overestimate the inventory of short-lived radionuclides by 8 to 33 percent. This is because in any reactor the inventory of isotopes with short half-lives that reach equilibrium during irradiation is approximately proportional to the power density (neutron flux). The inventory of radionuclides with long half-lives is essentially proportional to burnup (neutron flux times time); hence the calculation used underestimates the inventories of these isotopes by about 20 percent, inasmuch as current PWR's operate at about 33,000 MWD/MTU.

A comparison of boiling water reactors (BWR's) with PWR's can be made either on the basis of the entire core inventory or on a per MTU basis. For a BWR and a PWR of current design with the same rated thermal output, both the decay heat[2] and the radionuclide activity of the entire core would be approximately the same both for short and for long decay inventories. Second-order effects and minor differences in individual isotopes would exist and would be attributable to differences in enrichment, fuel geometry, in-core fuel management, power density, etc.

A comparison on a per MTU basis between the two reactor types depends principally on core size. For reactors of typical current designs with similar thermal outputs, the core sizes are 136 MTU for a BWR and 88 MTU for a PWR. The average power density of $\sim$26 MW/MTU for that BWR is considerably lower than that for the PWR ($\sim$38 MW/MTU). The average discharge burnup is also lower for the BWR than for the PWR: 28,500 MWD/MTU versus 33,000 MWD/MTU. These values are for specific examples of current designs of each reactor type.[3]

In general, the use of the PWR radionuclide inventory for modeling both reactor types will result in a conservative analysis. Changes in the calculated radionuclide inventory due to modifications and improvements in the code[3] have not been examined in detail, but are expected to be on the order of 5% or less for the isotopes which dominate the hydrospheric consequences.

A2.3 Nuclides Considered

The original list of radionuclides in the core inventory was reduced to 75 both on the basis of half-lives and on the basis of initial activities; nuclides with half-lives greater than one day and with initial activities within five orders of magnitude of the largest component were included. This list was narrowed to 53 on the basis of dominant dosimetry effects. This list was the one employed for instantaneous releases into surface waterbodies. A further reduction of the list on the basis of groundwater transport delays resulted in a list of 34 nuclides. This reduced list was based on a minimum one-day transit time for groundwater to the nearest surface waterbody. This list is given in Table A1. (The transit time for each radionuclide was obtained by using the expression for the average element velocity given in Section B.2 of Appendix B, combined with the values of the element distribution coefficients listed in Table A1.) One last reduction was performed by assuming that release of radionuclides not commence until at least one year after melt-through (see A3.4). The resulting list of 22 nuclides is also given in Table A1. Ten isotopes which are present in both lists in Table A1 are included only because of their presence in early parts of important decay chains. They are not expected to contribute significantly to the consequences themselves.

A3 Composition of the Melt

A3.1 Method

The interactions of the molten reactor core with the containment concrete and with the soil beneath the containment were taken to be adequately described by the models employed in INTER[4] and in WECHSL.[5] Most of the results presented in this appendix which relate to the descriptions of these interactions were obtained using the INTER code. However, new insights obtained using the more recently developed WECHSL code are also included. Except for purposes of determining the total decay heat, the detailed radionuclide composition of the reactor core was not considered for any of these calculations describing the core-reactor and the core-substrate interactions; rather, the radioactive portion of the core was approximated by 118 tons of UO_2.

The models employed in INTER incorporate the qualitative and limited quantitative experimental data available at the time of its development concerning core-reactor and core-concrete interactions. Among the phenomena included in the models are convective stirring of the melt by evolved gases, substrate decomposition, admixture of substrate decomposition products to the melt, core and structural material oxidation and reduction reactions, interaction with available water, radiative heat loss, and variations of heat transfer coefficients with local pressure. The models rely primarily on empirically-based descriptions and on estimates.

In the revised code, WECHSL, the empirical submodels and estimates used in INTER have been replaced by physically-based models. These models have been based on experiments with core-melt simulant materials and have been confirmed by experiments involving reactions of 50-100 kg of molten metallic and oxidic melts with concrete. Some of the major differences between the models behind each of the two codes are summarized in Table A2.

Table A1. Radionuclide Inventories

Nuclide	Initial Amount[a,b] (curies)	Amount After 1 Yr[c,d](curies)	Half-Life (days)	K_d
Co 58	7.78+5	2.23+4	7.13+1	75
Co 60	2.90+5	2.54+5	1.92+3	75
Ni 63	2.29+4	2.27+4	3.36+4	80
Sr 89	9.41+7	7.24+5	5.20+1	20
Sr 90	3.71+6	3.62+6	1.03+4	20
Ru 103	1.08+8	1.80+5	3.96+1	0&4
Ru 106	2.48+7	1.24+7	3.69+2	0&4
Ag 110m	1.02+5	3.75+4	2.52+2	250
Sn 123	6.30+5	8.31+4	1.29+2	250
Sn 125[e]	1.04+6	-	9.65+0	250
Sb 125	3.49+5	2.78+5	9.97+2	15
Sb 127[e]	6.08+6	-	3.90+0	15
Te 127m	1.13+6	1.11+5	1.09+2	20
Te 129m	5.28+6	-	3.34+1	20
Te 131m[e]	1.27+7	-	1.25+0	20
I 131	8.48+7	-	8.04+0	0
Cs 134	7.49+6	5.34+6	7.52+2	200
Cs 136	3.02+6	-	1.30+1	200
Cs 137	4.67+6	4.56+6	1.10+4	200
Ba 140	1.50+8	-	1.28+1	30
Ce 144	8.51+7	3.49+7	2.84+2	600
Nd 147[e]	6.05+7	-	1.10+1	600
Pm 147	7.74+6	6.49+6	9.58+2	600
Sm 151	6.69+4	6.64+4	3.39+4	600
Eu 54	1.79+5	1.71+5	3.14+3	600
Np 238[e]	1.49+7	-	2.10+0	15
Np 239[e]	1.64+9	-	2.35+0	15
Pu 238[e]	5.70+4	5.95+4	3.25+4	2000
Pu 239	2.13+4	2.17+4	8.91+6	2000
Pu 240	2.07+4	2.07+4	2.47+6	2000
Pu 241	3.36+6	3.20+6	5.33+3	2000
Am 241	1.65+3	6.90+3	1.67+5	2000
Cm 242[e]	4.96+5	1.05+5	1.62+2	600
Cm 244[e]	2.28+4	2.19+4	6.61+3	600

[a] Assumes immediate release at the time of melt-through followed by a minimum 1-day groundwater travel time.

[b] 7.78+5 denotes 7.78×10^5.

[c] Assumes a one-year delay after melt-through followed by a minimum 1-day groundwater travel time.

[d] If no amount is given, it is considered to be negligible and the radionuclide is not included in the list.

[e] This isotope is included only because of its presence in the early part of a possibly important decay chain.

Table A2. Differences Between INTER and WECHSL

Phenomenon	INTER	WECHSL
Heat Transfer to Concrete	Empirical Coefficients	Physical Model
Heat Transfer between Phases	Estimate	Physical Model
Void Fraction and Swell	Ignored	Physical Model
Geometry	Cylinder/Sphere	Accurate Geometry
Oxidation Reactions	Complete Reaction	Reaction to Equilibrium
Material Properties	Mixture Formula	(a)
Freezing Points	Estimate	Schroeder-van Laar[7]

[a] Viscosity and density are determined by the VISRHO[6] model; other properties are determined by mixture formulae.

Because of the number of assumptions and approximations which the basic models behind INTER and WECHSL incorporate, it cannot be supposed that the codes can be used as quantitative predictive tools. However, it is believed that the codes are quite useful for qualitative discussion of the major phenomena involved in the core-substrate interaction. In particular, the codes are at least useful for assessing the parametric dependences of many features of that interaction.

Replacement of the estimates and the empirical models of INTER by the physically-based models of WECHSL has reduced the band of uncertainty in the results. However, some important coefficients in the latter code (such as the emissivity of the upper melt surface) are still difficult to estimate with accuracy. Furthermore, some of the models in WECHSL, although more satisfactory than the estimates used in INTER, are still much simplified from the actual physical situation. For example, neither of the sets of models behind the two codes correctly considers the reaction of the melt debris with the concrete or soil after solidification.[*] Unfortunately, in determining the melt-through time, the very large uncertainties in the post-solidification rate are much more important than the minor uncertainties in the rate of penetration while the melt is still liquid. Therefore the band of uncertainty remaining in the melt-through time is large. In contrast, the volume of concrete or subsoil melted is nearly independent of the remaining uncertainties in the data.

A3.2 PWR Calculations

A3.2.1 Composition of the Melt in a PWR Before Contact with Concrete

Many possible accident sequences can result in containment melt-through. In order to keep the present study as consistent as possible with the RSS, the accident sequences of that study have been utilized. The reactor and containment designs considered in detail here are also those analyzed in the RSS.

The composition of the melt in its early stages will be very dependent upon the accident sequence. In a first approximation, it is possible to categorize the initial part of any meltdown sequence by two parameters: the time from accident initiation to contact of the melt with the concrete; and, the amount of water available to the melt. Delayed sequences with ample water will give rise to compositions differing from relatively rapid sequences with little water in the following particulars: more of the reactor steel will be incorporated; a greater degree of oxidation will exist; and a lower decay power will be present.

German investigators[8] have considered two melt compositions, "Corium A" and "Corium E". Corium A contains less steel than Corium E and is similar to what might be expected if the reactor pressure vessel (RPV) were melted through in a relatively short time (2 hours) after accident initiation. For several hours delay from accident initiation to RPV melt-through, the composition would more closely approach Corium E. These corium compositions are as follows:

Corium A	Corium E
118T UO_2	118T UO_2
33T Zircaloy-4	33T Zircaloy-4
31T Steel	171.5T Steel

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[*] Limited experimental data suggest that hot solids penetrate concrete at a rate of 2-3 cm/hr. Based on calculated solidification times, the time required to penetrate the concrete basemat is completely dominated by the post-solidification interaction.

Just before contact with the concrete, the melt for a relatively rapid accident sequence might include the contents of Corium A plus the bottom reactor head (30,000 lb. Fe) plus the lower liner (15,000 lb. Fe). For such a short duration melt sequence with an ample supply of water present (Sequence AD),[9] 75 percent of the Zircaloy[10] and 20 percent of both the core and the bottom head steels might be oxidized. For this situation, the following would represent the states of oxidation of the dominant strutural metals after RPV melt-through:

$Zr(\text{metallic}) = 33T \times .25 = 8.25T;$

$Zr(\text{as } ZrO^2) = 33T \times .75 = 24.75T;$

$Fe(\text{metallic}) = (.74) \times (31T) \times .2 + 15T \times .8 + 7.5T = 37.85T;$

$Fe(\text{as } FeO) = (.74) \times (31T) \times (.2) + 15T \times .2 = 7.59T;$

$Cr(\text{metallic}) = (.18) \times (31T) \times (.8) = 4.46T;$

$Cr(\text{as } Cr_2O_3) = (.18) \times (31T) \times (.2) = 1.12T;$

$Ni(\text{metallic}) = (.08) \times (31T) = 2.48T.$

The overall composition of this melt prior to contact with concrete is as given in Table A3.

For a melt of sequence of short duration but with limited water availability (Sequences AB, AH, ACH, TMLB′), 75 percent of the Zircaloy and 10 percent of both the core and the lower head steels might be oxidized. The initial structural composition of the melt would be the same as for the previous situation. The final composition of this melt is given in Table A3.

Just before contact with the concrete, the melt for a relatively slow sequence might include Corium E, the bottom head and the liner. Larger fractions of the structural materials would probably be oxidized for any slow sequence. Therefore, for such a delayed sequence with ample water (Sequence AG),[9] 80 percent of the Zircaloy and 50 percent of both the core and the bottom head steels might be reacted. For a delayed sequence with limited water (no sequence), 80 percent of the Zircaloy and just 20 percent of all the nonliner steels might be oxidized. The final compositions of both of these melts are just after RPV melt-through are given in Table A3.

At this stage, the melt resulting from any sequence could be expected to consist of two immiscible layers, a metallic one and an oxidic one.[*],[**] The composition of the melt with respect to certain metallic and oxidic components is depicted in Figure A1. Variations in the composition prior to contact with the basemat concrete are given both as a function of the time from accident initiation to concrete contact and as a function of the availability of water.

If a steam explosion occurred, some part of the melt might be scattered and might not take part in concrete melt-through. Scattered fragments of melt might contact concrete, but because of their high surface-to-volume ratio they would remain quenched and would not decompose appreciable concrete. The scattered fraction could vary from nearly zero to a large fraction of the core.

Calculations reported in the RSS[12] indicate that any amount of the core exceeding 4 percent could, under some conditions, interact energetically enough to cause containment failure. The calculations on which the failure probabilities of the RSS[9] were based used 100 percent of core mass, however. Even if 100 percent of core mass entered into a fuel-coolant interaction, it is unlikely that all this material would be so scattered as not to take part in significant core-concrete interactions.

Table A3. Composition of PWR Melt Before Contact with Concrete (in tons)

	Short Time, Ample Water	Short Time, Limited Water	Long Time, Ample Water	Long Time, Limited Water	Short Time, Ample Water, Steam Explosion	Long Time, Ample Water, Steam Explosion
UO_2	118	118	118	118	88.5	88.5
ZrO_2	33.44	22.3	40.12	35.7	25.1	30.1
FeO	9.76	4.87	91.25	36.50	8.28	70.85
Cr_2O_3	1.64	.82	22.57	9.02	1.23	16.93
Zr	8.25	16.5	3.3	6.6	6.19	2.48
Fe	37.85	41.6	78.46	116.53	33.26	65.29
Cr	4.46	5.02	15.44	24.7	3.34	11.58
Ni	2.48	2.48	13.72	13.72	1.86	10.29

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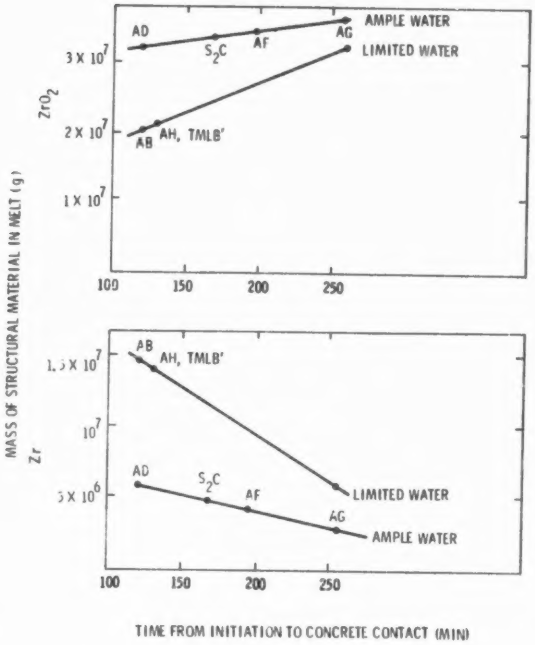
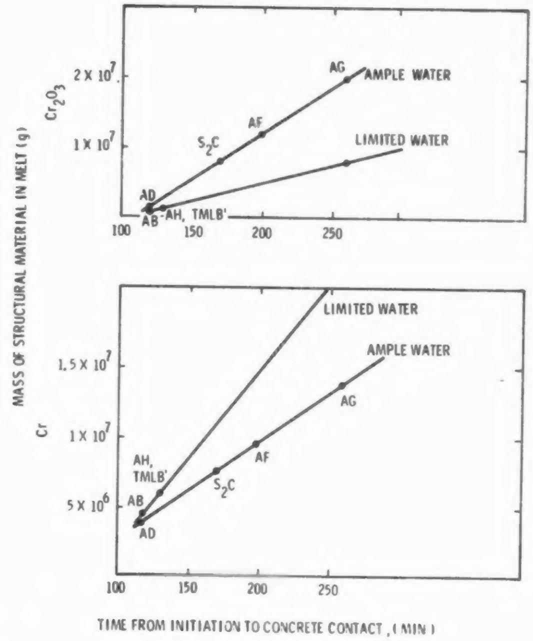
[*] Oxidation of the entire melt would proceed as the molten progressed through the concrete and soil. By the time the melt finally came to rest beneath the containment, most of the constituents would be present in the form of oxides. That is, all those metals which formed stable oxides would be present as such.

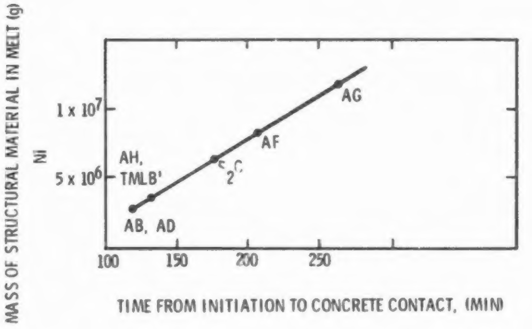
[**] Fischer, et al.,[11] have studied the distribution of fission products to metallic and oxide phases at early stages. They found that Y, La, Sr, Ba, Zr, Pr, and Ce were distributed to the oxide and Mo and Ru were distributed to the metal. Nb was split between the two phases. The distribution of the Cs-Rb and Te-Sb groups is split between metallic and oxide phases in this study. Ten percent of the lanthanide group (which includes Nb) is distributed to the metal. Other distributions follow the results of Fischer, et al.



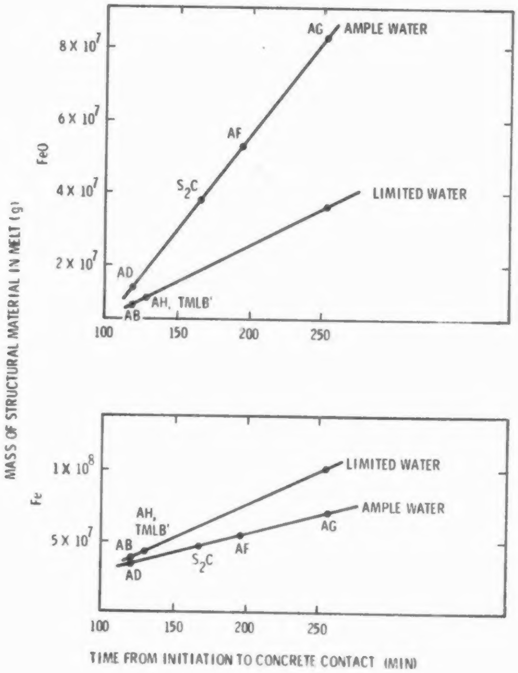
a. Zr and ZrO_2



b. Cr and Cr_2O_3



c. Ni and NiO



d. Fe and FeO

Figure A1. Metal and Metal Oxide Contents of Melt as a Function of Time After Accident Initiation

In this discussion, it is assumed that if a steam explosion occurred, then 25 percent of the core mass (at the time of interaction) would be permanently scattered out.* This scattered mass would include UO_2, ZrO_2, Zr, and core steel, but would not include the bottom head or the containment liner. The fraction scattered could be either higher or lower than 25 percent. Higher fractions would reduce the severity of the core-concrete interactions; it has been found that lower fractions do not have a significant effect.

Under steam explosion conditions, the makeup of the melts for a relatively rapid sequence with ample water and for a relatively slow sequence also with ample water, both with the assumption of 25 percent scattering, would be approximately as given in Table A3. It is improbable that energetic steam explosions would occur in sequences having limited water.

A3.2.2 Decay Power in the Melt

The heat generated by the melt is related to the net decay power of the radionuclides present in the melt. The loss of fission products during the course of the accident has been taken from the RSS.[9] The fraction of the decay power lost has been determined by assuming that the decay power of each nuclide, except for any actinide, is proportional to its activity; the power of actinides has been weighted by a factor of four relative to that of other nuclides. The reason for the higher actinide weighting is that many of them are α-emitters, which have much higher decay energies than β- or γ-emitters. Decay power after steam explosions has been further reduced by 25 percent to account for material scattered out of the reactor cavity.

ECCS acceptance criteria require the use of an ANS standard[13] on decay power plus a 20 percent addition for "uncertainty." More recent data[14] have shown that the standard itself, without the 20 percent addition, is adequate. Accordingly, in this study, fits to the ANS standard have been used.

A3.2.3 Interaction of the Melt in a PWR with Concrete

After the molten core melts its way through the reactor vessel in a PWR (Figure A2[15]), it drops into the reactor cavity immediately below. This cavity confines the melt to a relatively small region with respect to the area of the entire containment floor. If the containment design is like that in Figure A2, breaching of the shield wall can occur during melt-through. In this study, melt-through is assumed to proceed through the bottom of the reactor cavity.

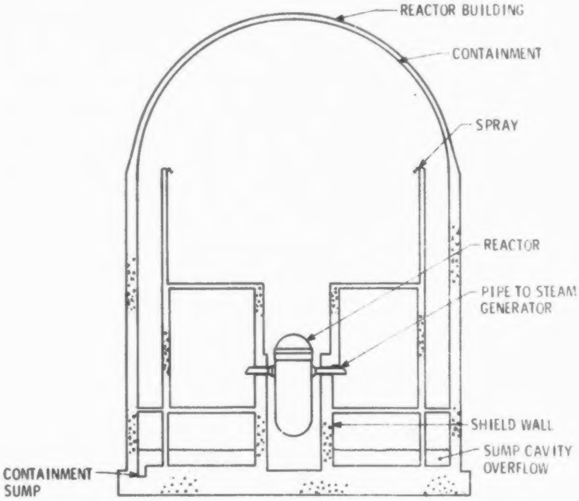


Figure A2. Cross Section Through the Reference PWR.

In order to consider the interactions of the molten core with the containment concrete, two types of concrete have been employed: carbonate aggregate and siliceous aggregate. The compositions assumed are given in Table A4. There is actually a continuous spectrum of concrete compositions of which the two types chosen for study effectively represent extremes.

Table A4. Concrete Compositions[16]

Material	Siliceous Weight Percent	Carbonate Weight Percent
SiO_2(a)	55.7	15.3
Al_2O_3(b)	20.9	3.3
$CaCO_3$(c)	0.2	64.9
$Ca(OH)_2$	21.6	12.7
H_2O	1.6	3.8

[a] Includes TiO_2, Na_2O, K_2O
[b] Includes FeO
[c] Includes $MgCO_3$

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*Limited experimental evidence seems to suggest that the probability of an ex-vessel steam explosion which would not damage the containment but would scatter core material is nearly unity. If steam explosions nearly always occur and if large fractions of the core are scattered so that concrete penetration cannot occur, the probability of subsurface releases could be reduced by an order of magnitude or more. Experimental data are still too scanty to confirm that this is indeed the case. The present study uses steam explosion estimates from the RSS, and is therefore, probably conservative.

As was indicated previously, the possible accident sequences can be classified with respect to both time before RPV melt-through and availability of water. Three of the four endpoints of these possible meltdown sequence ranges can be represented by sequences AD, AG, and AB (the RSS[9]). These sequences can be further differentiated by type of containment failure and hence containment pressure. The categorization is shown in Table A5. Note that no sequences give delayed contact with full water availability.

Table A5. Melt-Through Categories

Sequences	Contact With Concrete	Water Available	Pressure
AD-ε	Rapid	Ample	High
AD-β	Rapid	Ample	Low
AB-α	Rapid	Limited	Low
AG-δ	Delayed	Limited	Low

The amount of water available could be of importance if the shield wall between the PWR reactor cavity and the rest of the containment were breached. Water on the containment floor could then contact the melt and enhance the cooling of the melt.

The design of the plant selected was that used in the Reactor Safety Study,[12] with design parameters being abstracted from the appropriate FSAR.[15] The INTER code[4] was used for all concrete interaction calculations, unless noted otherwise. Melt-through times and other results for silica concrete are shown in Table A6.

The calculated time until melt-through occurs is 72 hours for all sequences considered. This is because the melt-through times are completely dominated by the post-solidification interaction, which is essentially the same for all sequences. Similarly, the shield wall is found to be breached for all accident sequences considered; however, calculations with slightly higher heat transfer from the metallic constituents and slightly lower from the oxidic than those used for these calculations show the shield wall surviving. On the basis of INTER calculations, it is believed that until the heat transfer coefficients can be more accurately determined, a probability of shield-wall penetration of about 50 percent should be considered reasonable.

Table A6. PWR Melt-Through Times

Sequence	Time to Contact Concrete (hr)	Time to Melt-Through (hr)[a]	Shield Wall Breached?
	Silica Concrete		
AD-ε	2.3	72	no[b]
AD-β	2.3	72	yes
AB-α	2.3	72	yes
AG-δ	27.2	72	yes
	Limestone Concrete		
AD-ε	2.3	124.2	yes
AD-β	2.3	97.7	yes
AB-α	2.3	72	yes
AG-δ	27.2	(220)[c]	yes

[a] The following values have been assumed: containment floor thickness, 10 ft; shieldwall thickness, 4 1/2 ft; penetration rate after solidification, 3 cm/hr.

[b] The shield wall is breached according to an INTER calculation; breaching is uncertain according to WECHSL calculations. Melt-through times are the same for all calculations. Breaching of the shieldwall is very sensitive to certain parameters of the calculation. Calculations have not been performed with the WECHSL code for the other sequences.

[c] Best estimate calculations with INTER show no penetration. However, penetration could occur as shown within a reasonable range of the input parameters.

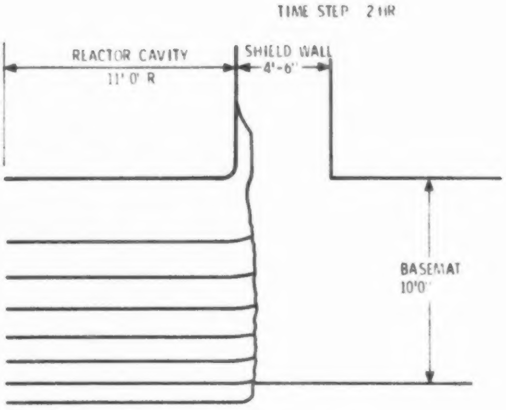
Limited recomputation of containment melt-through has been carried out with the WECHSL code. Sequence AD-ϵ was recomputed, and the parameters to which the results were most sensitive were investigated. It was found that the melt-through time was nearly identical to that given by the INTER calculations. There is no reason to expect significant differences for other sequences. Consequently, the INTER melt-through times are considered to be reasonably representative.

In contrast, WECHSL results for shield-wall breaching differed somewhat from those generated by INTER. The probability of shield-wall failure for the WECHSL calculations was very sensitive to the freezing point of the oxide phase. The reason for this sensitivity is that lateral growth of the cavity is controlled almost exclusively by the oxide layer. After solidification of the oxide phase, lateral cavity growth should become very slow.

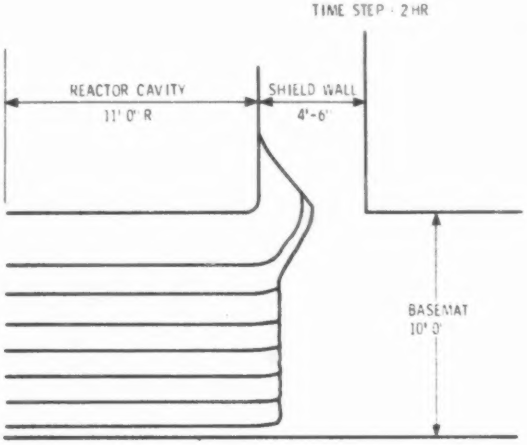
Figure A3 shows erosion contours (generated by WECHSL) as a function of time for high (Case I), medium (Case II) and low (Case III) freezing temperatures for the melt. The freezing temperatures for these cases were calculated by the method of Reference 7, using slightly different assumptions for each. The differences in freezing temperatures are functions of compositions. The freezing temperature of Case III is about 200°C below Case I.

Only a small fraction of the shield wall is melted in Case I (Figure A3a) and sumpwater release would be improbable. Somewhat less than 50 percent of the thickness of the shield wall remains intact in Case II (Figure A3b). Approximately 75 percent of the thickness of the shield wall is eroded away in Case III (Figure A3c).

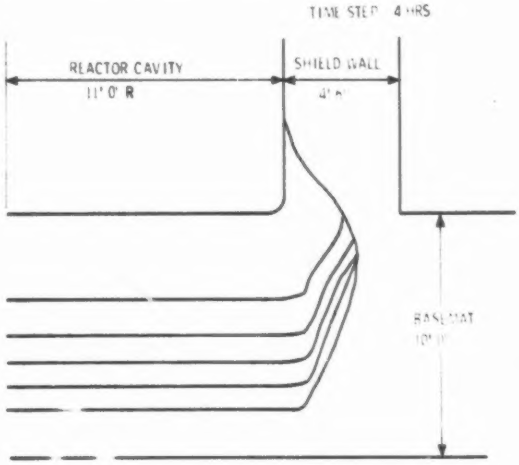
Case I, with very early freezing and little lateral growth, is believed unrealistic.*[17] Cases II and III, with noticeable lateral growth, are thought to be more likely.



a. For a Relatively High Freezing-Point Melt



b. For an Average Freezing-Point Melt



c. For a Relatively Low Freezing-Point Melt

Figure A3. Cavity Contours

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*Experiments have been conducted on crust formation in slags as a function of "foreign" oxide addition.[17] Even small admixtures of lower melting-point oxides were found to prevent crust formation or freezing; the lowering of the freezing point was greater than would be expected from binary phase diagrams. In view of the known tendency for iron oxides and molten concrete constituents to effect very great reductions in freezing points, Case III is considered most probable.

In Cases II and III, sizeable fractions of the shield wall are eroded, and the remainder will be thermally altered and thermally stressed. A precise quantification of the probability of shield-wall failure is not possible. Although total shield-wall collapse appears unlikely, major cracking is possible. In agreement with the INTER calculations, it is estimated that the probability of cracking or local collapse, sufficient to admit a sizeable flow of sumpwater onto the melt, is of the order of 50 percent. It must be borne in mind that this estimate is subject to major uncertainties, and could be subject to considerable revision in view of variations of design, input parameters, or accident sequence. For example, a single calculation of a different reactor design with an improved version of WECHSL showed more extensive shield-wall melting.

The results of INTER calculations for a limestone concrete are also shown in Table A6. The time until melt-through occurs varies from 72 hours to infinity. As for the silica concrete, the shield wall is breached for all accident sequences considered, according to INTER calculations. Complete penetration of the limestone concrete basemat does not occur in sequence AG. It should be pointed out, however, that penetration would be achieved with slightly higher heat transfer coefficients and lower heats of decomposition. These results are also dependent on the aggregate composition. Accordingly, a probability of penetration of 50 percent is assigned for those sequences, with a time of penetration of 220 hours.

A3.2.4 Composition of the Melt from a PWR After Contact with Concrete and Soil

The penetration of the soil underneath the containment has also been computed using the INTER code. Analogously to the consideration of concrete melt-through, the earth penetration problem can be bounded by considering the siliceous concrete to be underlain by a layer of siliceous gravel, and the carbonate concrete by a layer of carbonate gravel. (See Chapter 2.) Therefore, these two limiting cases have been considered in this report.

The molten masses are found to behave similarly for all accident sequences in siliceous concrete and soil. That is, the concrete penetration times and the final compositions are essentially independent of sequence. Therefore, sequence AD-ϵ was used as a proxy for other sequences. The final composition of the siliceous melt is given in Table A7.

Table A7. Final Composition of Melt for a PWR

Compound	Silica Concrete/ Silica Gravel (wt. %)	Limestone Concrete/ Limestone Gravel (wt.%)
SiO_2	85.9	8.3
CaO	1.7	70.7
Al_2O_3	6.4	6.0
UO_2	2.4	6.5
FeO	2.2	5.3
ZrO_2	1.0	2.7
Cr_2O_3	0.3	0.4
NiO	0.2	0.1

As long as the mass were molten, it would continue to incorporate concrete and soil residua into itself. The calculations show the mass alternately molten and frozen. When the mass is molten, gases passing through it keep it well stirred; at such times, the heat transfer from the mass is high. As a consequence, the mass chills and freezes. However, the heat transfer from the frozen mass is lower, and becomes lower yet as the cooler frozen boundary layer becomes thicker. There comes a time when the heat transfer out of the mass is less than the decay heat, and the interior commences to heat up and eventually the mass remelts. The model for remelting and reestablishment of gas stirring in the INTER code is rather simplistic, so it is doubtful that this oscillatory behavior actually occurs. More likely, the mass adopts some semi-molten state with just sufficient circulation so that outward heat transfer balances the decay heat. At any rate, the results should not be dependent on the fine details of cooling.

The computer time required to follow the solidified mass to a relatively cool temperature would be excessive. Furthermore, the models for post-solidification behavior are not well founded. However, the time at which cooling occurs can still be estimated. (The losses from the frozen mass, at a time when the center is below the melting point, can be equated to the decay heat. The approximation of Shure[16] can then be used to determine the time at which decay heat will just balance the losses.) By six months after melt-through, the decay heat could probably be balanced by conduction with the interior still frozen. At later times, the mass would be solid throughout.

It must be pointed out that the models in both INTER and WECHSL are only applicable to molten pools. Both sets of models are based on experiments in which the pool temperature is significantly higher than the decomposition temperature of the surrounding medium. Post-solidification behavior as given by the models is no better than a simple heat balance. On the other hand, it has been experimentally determined[17] that delay in solidification and reliquefaction are both very important phenomena in oxide pools, so that some part of the melt may remain at least partially molten at relatively low temperatures.

In view of the many uncertainties in the descriptions of some of the phenomena, and because of inapplicability of many of the models, it should not be surprising that the later time behavior cannot be stated precisely. The estimate for the cooling time should be associated with very large uncertainties; even the width of the related uncertainty band is unknown.

Because neither model is properly applicable to the late time behavior, WECHSL calculations have not been carried out for the penetration of the soil under the basemat. There would be no reason to expect that WECHSL calculations could reduce the uncertainty in the INTER results.

As has already been mentioned, earth penetration has also been computed for a containment with a limestone concrete basemat. The material underlying the containment was taken to be a limestone gravel. Unlike the calculations for a siliceous substrate, the penetration time and ultimate composition are very much dependent on accident sequence. Penetration is much delayed for sequences in which either the containment pressure is above atmospheric* or the water in the containment sump can contact the melt.

As for the siliceous concrete and soil case, it would require excessive computer time to follow the melt from each sequence through the soil. Sequence AD-β will be taken as a "typical" or average sequence for earth penetration. The final composition of the limestone-based melt is given in Table A7.

A3.3 BWR Calculations

A.3.3.1 Composition of the Melt in a BWR Before Contact with Concrete

Although it is possible that the composition of the molten core prior to contact with the containment concrete is slightly different for meltdown accidents in BWR's and PWR's, the differences cannot be established with sufficient certainty to warrant the additional calculations required. It is to be expected that the BWR melt would have a somewhat higher metallic content than would a PWR melt; however, this is completely overshadowed by the significant differences in containment design. Accordingly, an initial melt composition identical to that for PWR sequence AD (Table A3) has been used for rapid BWR melt sequences, and an initial composition identical to that for sequence AG (also Table A3) has been used for delayed sequences.

A3.3.2 Interaction of the Melt in a BWR with Concrete

The containment design of a BWR (Figure A4[20]) is very much different from that of a PWR plant. In a core-melt accident, the molten core would initially drop into the cylindrical, steel-lined reactor support cavity. If the three-foot thick reactor support walls were breached, the melt-through could take one of several paths:

1. If the melt's upper surface were not significantly above the floor level, the melt could remain confined at about the outside radius of the reactor support cavity.
2. The melt could spread out to the radius of the primary containment and continue to melt downward from the new configuration.
3. The melt could flow into the vapor suppression system pipes. These pipes would promptly melt, allowing the molten mass to drop to the floor of the vapor suppression chamber.
4. The molten mass could enter the vapor suppression pool and be quenched.

Initially, the melt would always be contained within the reactor support cylinder. This cylinder could contain water up to a two-foot depth; the melt-through time of the support cylinder walls would be somewhat dependent on the presence of such water (Table A8). In contrast, the presence of water in the drywell outside the reactor support cylinder would make little difference in melt-through time; the configuration of the melt would not allow efficient cooling unless fragmentation took place.

After breaching the support walls, the melt would spread out across the drywell floor. At that point, the level would be too low to reach the drywell vent pipes. The resulting configuration is shown in Figure A5. For calculation, the configuration has been simplified to that shown by the dashed line shown in that figure.

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*This is so because the decomposition temperature of calcite is pressure dependent. At high pressures, calcite decomposes at a higher temperature than can be sustained by the melt.

Table A8. Melt-Through of BWR Side Walls

Sequence	Water Present	Time of Melt-Through After Start of Concrete Reaction[a]	Height of Melt Above Floor at Time of Wall Melt-Through	Height of Melt When Spread Out Across Dry Well Floor
TC α	Yes	5.4 hr	73 cm	26.0 cm
TC α	No	4.5 hr	74 cm	26.5 cm
TW α	Yes	Melts through bottom before melting side walls.		-
TW α	No	Melts through bottom before melting side walls.		-

[a] The following values have been assumed: containment floor thickness, 10 ft; sidewall thickness, 2.5 ft.

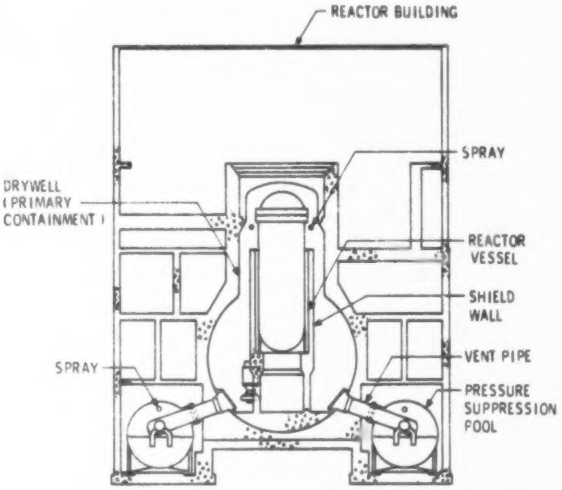

Figure A4. Cross Section Through the Reference BWR.

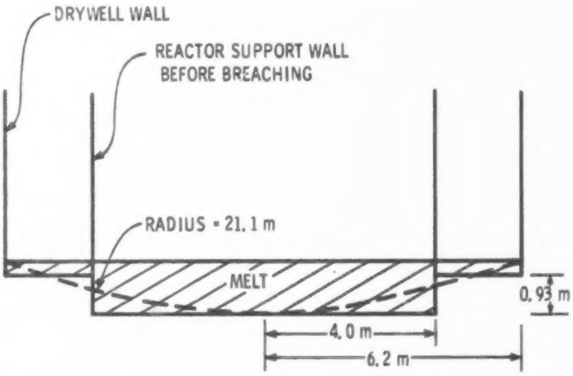

Figure A5. Configuration of Melt in BWR After Side Wall Melt-Through

As explained in the RSS,[4] loss-of-coolant-accident (LOCA) sequences do not make an appreciable contribution to the accident probabilities for BWR's. Two accident sequences completely dominate the picture: sequence TC, involving failure to make the reactor subcritical following a transient event; and sequence TW, involving failure to remove decay heat following a transient event. In both sequences, atmospheric pressure prevails throughout the melt-through. An additional, less probable sequence, TQUV, has also been analyzed to determine the effect of elevated pressure.

Melt-through times for a BWR with a siliceous concrete basemat are shown in Table A9. These melt-through times are dominated by the penetration rate of the solidified debris, as in a PWR. The average time of 72 hours is not significantly affected by either the accident sequence or the concrete type.

Table A9. BWR Melt-Through Times

Sequence	Time to Contact Concrete (hr)	Time to Melt-Through Silica Concrete Basemat (hr)
TC	3.3	72
TW	32.8	72
TQUV	-	72

A3.3.3 Composition of the Melt in a BWR After Contact with Concrete and Soil

The final composition of the mass resulting from melt-through of siliceous concrete into siliceous soil beneath a BWR is given in Table A10. It will be observed, by comparison with Table A7, that the composition of the final melt is almost identical to that for PWR melt-through of siliceous concrete and gravel. In order to save computer time, it was assumed that, in all circumstances, the cooling times and compositions for BWR melt-through are identical to those for PWR melt-through. It is believed that any minor differences will be thoroughly over-shadowed by differences in concrete and soil compositions from those assumed in this study and by the very great uncertainties in the models.

Table A10. Final Composition of Melt for a BWR

Compound	Silica Concrete/Silica Gravel (Weight %)
SiO_2	87.7
Al_2O_3	5.8
UO_2	2.2
FeO	2.2
CaO	1.6
ZrO_2	0.3
Cr_2O_3	0.0
NiO	~0

A3.4 Cooling of the Melt by Groundwater

One major source of cooling of the melt from either a PWR or a BWR will be contact with the groundwater beneath the containment. A major uncertainty in the consideration of this cooling applies to the migration of water within the gravel. Calculations indicate that the melt will be quenched almost immediately if the gravel is saturated and the groundwater is free to contact the hot melt. The calculations reported in the addendum of this appendix indicate, however, that only a fraction of the water will migrate into the melt. If the containment pressure is above the pore pressure, no water will migrate inward and the melt will be interacting with essentially dry soil. In such a case, the cooling of the melt will be delayed.

Experiments conducted under the Molten Core-Concrete Interaction Program[19] have shown that some of the free water in a concrete crucible containing molten prototypical core material migrates to the exterior of the crucible and some remains to be vaporized by the molten core. The migration of water in concrete is believed to take place along fine cracks. The phenomenon is believed to be similar to migration of water through soil.

On the basis of models and calculations discussed in the addendum,[21-23] it appears that some fraction of the vaporized water will always be incorporated into the melt. The fraction is, of course, larger if the pore pressure exceeds the containment pressure, but the model is not adequate to determine the fraction accurately.

Because of the lack of experimental information which can be used to determine the fraction of water vapor escaping to the vicinity of the melt, arbitrary fractions of 25 percent and 50 percent have been used. Variation of these fractions affects the cooling time of the mass. Several calculations have been performed with the INTER code using different fractions. It has been found that final composition of the mass is not greatly affected, although the cooling time can be changed by more than a factor of two.

On the assumption that 75 percent of the soil water migrated away from the melt without vaporization, continuous cooling would be obtained in 3.2 years. On the other hand, if it is were assumed that 50 percent of the soil moisture is evaporated by the melt, such cooling could be reestablished in 1.2 years.

It is to be expected that a more intimate contact might be established between the frozen mass and the soil after soil decomposition ceased. Of course, this would increase the heat loss from the mass, allowing more rapid cooling.

For the foregoing reasons, the calculated cooling time of 3.2 years is believed to be too long, and a time of two years to continual cooling is believed more reasonable. This falls within the estimate of Ritzman, et al.,[24] who have calculated a time of one to two years.

At the time that continuous cooling commenced, the surface temperature would be about 500°C. If significant groundwater flow were established at this time, cooling could be much more rapid. Initially, only the outer surface would be cool enough for water to contact it without boiling and cooling would still be relatively slow. However, cooling would be greatly accelerated as soon as groundwater could penetrate farther into the debris. The cooling times for melts of differing compositions would be expected to be comparable, provided there were equal access to groundwater for cooling.

It has been assumed in the computation of these cooling time estimates that groundwater would not contact the melt before complete solidification had occurred. However, it might not be necessary for complete solidification to occur before groundwater could contact at least the peripheries of the melt. Inasmuch as the melt is not expected to be very thermally conductive, such contact would probably not significantly increase the overall rate of cooling of the melt.

A4 Releases to the Hydrosphere

A4.1 Depressurization Release

A depressurization release is defined to be the sudden release of a part of the atmospheric contents of the containment into the underlying soil. It can occur only if the containment pressure exceeds the soil pore pressure at the time of melt-through. This condition is not met for any accident sequence in which the containment fails initially by means other than melt-through; that is, there can be no depressurization release if any containment failure occurs prior to melt-through. Depressurization can be either partial or complete, depending upon whether or not the containment sprays operate.

If the sprays do not operate, then "complete" depressurization can occur. The dominant PWR sequences clearly meeting the conditions for complete depressurization are:

AB-ϵ $(P = 1 \times 10^{-9})$
S_2HF-ϵ $(P = 1 \times 10^{-9})$*
TMLB′-ϵ $(P = 6 \times 10^{-7})$

Such release from PWR's is dominated by sequence TMLB′-ϵ. Thus, the probability of complete depressurization will be taken to be the probability of that sequence (Table 2.1 in Chapter 2). The RSS estimated a probability of meltdown (per reactor year) of approximately 4×10^{-5}.** Thus, it appears that clear cut, nearly complete, depressurization release can only occur in two percent of all meltdown accidents.†

With at least partial operation of the sprays, water vapor will be condensed. However, noncondensible gases can still cause some elevation of the containment pressure. The probability of a partial depressurization release for a PWR is taken to be (probability of meltdown) x (probability that containment remains intact until melt-through), or approximately 1×10^{-5} per reactor year (Table 2.1).

In BWR's such as those considered in the RSS,[20] containment failure is assured prior to melt-through. As a result, depressurization failure can never occur in such BWR's. It perhaps could take place in BWR's of more recent design.

In any depressurization release, the containment contents will only exhaust into the soil until the containment pressure is reduced to the soil pore pressure. The fraction of containment atmosphere escaping into the soil (assuming adiabatic depressurization) will be

$$f = 1 - \left(\frac{P_s}{P_1}\right)^{1/\alpha},$$

where P_s is the soil pore pressure, P_1 is the containment pressure at the time of melt-through, and α is the ratio of specific heats.

The soil volume in which the particles and gases are filtered from the atmosphere can only be estimated. Reference 25 contains a compilation of information on filtration of aerosols by sand and soil. The efficiency of such filters decreases with increasing velocity. However, the flow of gases through the soil with only the containment pressure driving the flow would certainly be low.

Let it be assumed that the soil removes a constant fraction of the remaining particles per unit length of soil traversed; that is, $dc/dx = -kc$ where k is a constant and c is the local concentration. In one set of experiments quoted,[25] a 7.6 cm column of sand removed 80-90 percent of 0.3 μm particles. If k is derived from these experiments, then the simple model indicates that the concentration would be reduced to 10^{-4} of the containment concentration in only 43 cm of such sand. Even allowing for the crudeness of the model, and considering the possibility that the experiments quoted are not strictly applicable to depressurization release conditions, it seems certain that virtually all the containment contents vented into the ground will be deposited initially within 10 m.

The release into the ground will actually occur over some extended time owing to the finite permeability of the subsoil. During this time, natural deposition processes within the containment will be occurring which will reduce the concentration of particles expelled. An opposite effect will be caused by gas production after melt-through which will tend to increase the fraction of containment contents expelled. Neither of these effects is susceptible to precise calculation. However, because they have opposite results and should both be small, they will be ignored.

Values for containment atmosphere contamination at the time of melt-through have been taken from Reference 26 and are presented in Table A11. The upper and lower bounds in that table correspond to estimated bounds on the melt-through time. Two sequences, AB-ϵ and AD-ϵ, have been considered. AB-ϵ is considered to be typical of those sequences in which containment sprays do not work (i.e., complete depressurization release), while AD-ϵ is typical of those in which containment sprays do work.

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*For the plant analyzed in the RSS, the probability for sequence S_2HF-ϵ was approximately 1×10^{-9}. For some other plants, in which ECR and spray recirculation failure are closely coupled, the probability could be two orders of magnitude or more higher.[36] Therefore, a probability of 5×10^{-8} may be more appropriate for sequence S_2HF-ϵ for a "generic" plant.

**See footnote in Subsection 4.3.

†This is based on the consideration of only the Reference PWR considered in the RSS. Detailed consideration of other reactors could change this value substantially.

Table A11. Fraction of Core Inventory Airborne at Melt-through(a)

		Xe	I	Cs	Te	Sr	Ru	La
AD-ε	max	1.0	1.3×10^{-3}	1.9×10^{-6}	5.6×10^{-6}	1.6×10^{-7}	3.5×10^{-7}	6.7×10^{-8}
	min	1.0	3.0×10^{-5}	~0	~0	~0	~0	~0
AB-ε	max	1.0	1.0×10^{-2}	2.1×10^{-1}	2.5×10^{-1}	2.3×10^{-2}	1.9×10^{-2}	3.2×10^{-3}
	min	1.0	1.0×10^{-2}	2.0×10^{-2}	2.0×10^{-2}	2.2×10^{-3}	1.6×10^{-3}	2.6×10^{-4}

(a)See Table A13 for a listing of the other elements in each of the 7 chemical classes considered here.

The internal pressure for sequence AB-ε is estimated to be 76 psia, and the pressure for sequence AD-ε is estimated to be 59 psia.[26] An external pressure of 34 psia is assumed as was done in Reference 9. For these pressures, the fractions of containment atmosphere released are:

AB-ε f = .44

AD-ε f = .33

These fractions are applied to the containment atmosphere descriptions given in Table A11[26] to give the fractions of core inventory released to the soil as a result of depressurization.

The fractions calculated for depressurization releases are dependent principally on the ratio of containment pressure to pore pressure; neither pressure is well defined. Also, as was mentioned above, the competing processes of fallout and continuing gas production could alter these estimates. Thus the depressurization release fractions must be considered only representative of possible conditions. It should be understood that the actual depressurization release fractions could be either higher or lower than the quoted figures.

A4.2 Sumpwater Release

There is a possibility in PWR's of some designs that part of the contaminated sumpwater can be released into the ground through the pathway formed by the melt. Like depressurization, sumpwater release occurs relatively rapidly. In sequences in which the containment sprays work, the principal fraction of radioactive material released from the core can be contained in the sumpwater. Therefore, even if the probability of sumpwater escape were small, the risk to the public could conceivably be larger than for other escape modes.

Sumpwater can escape from a PWR of design similar to that described in Reference 15 if the shield wall between the reactor cavity and the rest of the containment is breached. That probability has previously been estimated to be about 0.5 (Subsection A3.2.3).

In addition to breaching of the shield wall, such sumpwater escape also requires that adequate water be available to enter the containment sump. Sequences such as AB, AH, or TMLB′ do not meet these conditions.

If the containment has failed prior to melt-through, water contacting the hot melt will probably flash into steam and will be lost from the containment. Thus, all sequences in which initial containment failure occurs other than by melt-through could most likely be ruled out.

The dominant sequences remaining for sumpwater escape into the ground are AD-ε, S_1D-ε, and S_2D-ε. Insignificant contributions are made by sequences AHF-ε, ADF-ε, S_1HF-ε, and S_2HF-ε.* As a result, the probability of sequences in which sumpwater escape is possible by shield-wall breaching is very nearly 10 percent of all LOCA's.

Figure A6 depicts qualitatively possible steps in a PWR sumpwater release by shield-wall breaching. Figure A6a shows conditions immediately after melt-through. The shield wall has failed and the sumpwater has been admitted to the top of the melt. Evaporation and recondensation of the sumpwater keep the water level nearly constant. The upper surface of the melt is rapidly frozen as its heat is delivered to evaporate the overlying water.

Figure A6b shows the situation after the melt has stopped growing but before it has solidified. The plug of material remaining in the hole melted through the basemat has solidified, cooled, and shrunk away from the wall. Experiments indicate that this remaining plug will be severely cracked and possibly porous, so that many paths will exist for sumpwater penetration. When the mass has cooled below the Leidenfrost temperature* (~500°C for oxides), heat transfer to the water is enhanced, and cooling becomes much more rapid.

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* This is true for the plant analyzed in the RSS. However, for the "generic" plant, in which the probability of HF type accidents is larger, sequences AHF-ε, S_1HF-ε, and S_2HF-ε make significant although not dominant contributions.

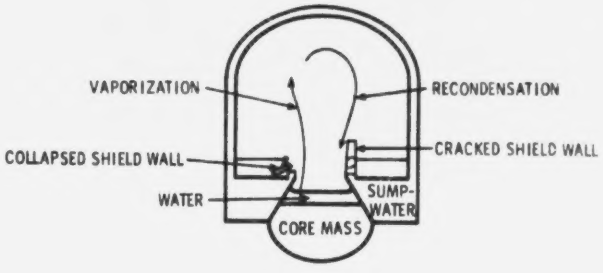
In Figure A6c, the upper portion of the mass has cooled sufficiently to allow sumpwater to penetrate to the underlying soil. Sumpwater can now escape propelled by the gravity head plus the containment pressure.

The time of escape might be considerably delayed by the above-mentioned boiling and recondensation. It is improbable that leakage in any significant amount would occur while active growth of the melt were taking place. This growth has been estimated to stop after about one month. Thereafter, leakage could be fairly rapid, with the actual rate depending on both the soil permeability and the containment pressure. The latter should be quite low - probably well below the soil pore pressure - at such a long time after the accident. With low pressure, there would be no mechanism for actively forcing the sumpwater out into the soil, and mixing with groundwater could occur only as a result of entrainment by flowing groundwater.

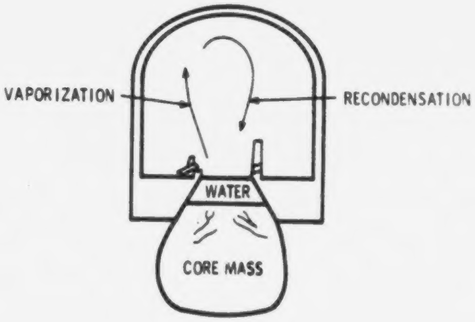
Sumpwater escape by shield-wall breaching is virtually impossible in some reactor designs. For example, it is difficult to see how sumpwater could escape from typical ice-condenser type plants. However, a probability of sumpwater escape of 5×10^{-6} is assumed for all PWR's, based on the probability of sumpwater release by shield-wall breaching in plants like those in Reference 15 (Table 2.1).

It is theoretically possible for a PWR sumpwater release to occur without shield-wall breaching if emergency core recirculation (ECR) continues for a long enough period after core melt. As was indicated previously, the release of sumpwater is expected to be delayed for days or weeks after accident initiation. If ECR were continued for an indefinite period after core melt, the sumpwater could eventually be permitted to escape. However, it is believed that common sense will dictate that the Emergency Core Recirculation System (ECRS) be shut off within a few days after core melt.

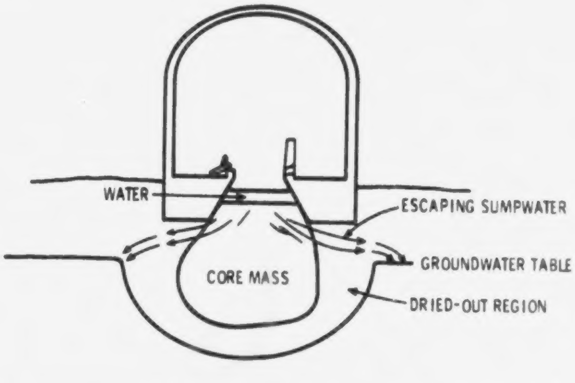
One could estimate an upper bound of the probability of such sumpwater release by ECRS operation by considering those sequences in which the ECRS is operable and the shield wall is not breached and by assuming that ECRS operation continued indefinitely. However, in view of the uncertainty in shield-wall failure, this procedure is deemed to be an unnecessary refinement.



a. Approximately 3 Days after Accident Initiation: the Sumpwater Leaks through Collapsed Shield Wall onto the Melt; Vaporization of the Sumpwater Occurs.



b. Approximately 1 Month after Accident Initiation: the Sumpwater Partially Penetrates into Openings at the Top of the Melt and is Evaporated, Further Cooling the Top of the Melt; Condensation Recycles the Vapor.



c. Approximately 6 Months after Accident: the Upper Part of the Core Mass has Cooled Sufficiently, Permitting the Escape of Sumpwater into the Ground.

Figure A6. Steps in Hypothetical Sumpwater Escape

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*The Leidenfrost temperature is that temperature above which a vapor shield is formed.

BWR's do not have containment spray systems analogous to those found in PWR's. Consequently, even if leakage of water into the ground were to occur from a BWR, it would not be expected to release a significant amount of radioactivity to the soil. Its primary effect would be to hasten cooling of the melt. The impacts of such releases are not considered further.

The fraction of core inventory initially contained in the sumpwater could be as high as the fraction released from the melt minus the fraction remaining in the containment atmosphere at the time of melt-through. However, not all of this radioactivity would necessarily be contained in the sumpwater which eventually escaped. In cases in which the sprays worked continuously, any material settling on other surfaces (other than that plated out within the primary system) could be assumed to be eventually washed off and picked up by the sumpwater. However, in any sequence where containment spray recirculation failed, some material would undoubtedly be left behind. In addition, some sumpwater could remain in sumps, piping, and so forth. The fraction remaining would be very much dependent on the details of the shield-wall failure and the plant design. No method is known for estimating the amount lost from the sumpwater by the various possibilities, but it could be a large fraction. If the watertable were higher than the bottom of the containment base, groundwater could eventually seep in and redissolve some of the soluble material left behind by evaporating water or plate-out.

In view of the many uncertainties involved, it is estimated that a conservatively large fraction of sumpwater - perhaps 50 percent - could conceivably escape. As was mentioned previously, a large portion of the radioactivity removed from the containment atmosphere by the sprays could be with that sumpwater. Therefore, it might be assumed that the sumpwater fraction would be very nearly equal to the amount volatilized from the core except for the noble gases and organic iodides. However, it is much more likely that some of the more volatile nuclides, such as I, Cs and Te, would be returned to the atmosphere during the period of boiling and recondensation. These nuclides could possibly leak from the containment or could settle on containment surfaces. It is thus believed likely that a considerably smaller fraction of some of the more volatile nuclides -- say, 10 percent -- would be available for release via the sumpwater escape.

Because of the long time expected before appreciable amounts of sumpwater could escape, it is considered likely that ameliorative measures could be taken. If the Containment Spray Recirculation System (CSRS) were available, it might be relatively simple to pump out the sumpwater to some temporary storage area. If the CSRS were unavailable for a substantial period of time, then the most probable sequences could be ruled out. The dominant sequence for which the CSRS would be inoperable for an extended period is $S_2HF\text{-}\epsilon$, having a probability of 1×10^{-9}.[*] It has already been estimated that the probability of the shield wall being breached is of the order of 50 percent; therefore, the combined probability of sumpwater escape without the possibility of ameliorative measures is about 5×10^{-10}.[**]

It should be noted that the sumpwater release considered in this appendix, that is, release directly to the substratum at the reactor site, is not the only possibility. Sumpwater might conceivably escape from the reactor complex by some route other than through the pathway formed by the melt. Such escape might even result in immediate entry into readily accessible portions of the hydrosphere. The probability of such an escape is presumably not negligible but it is difficult to quantify. It could be upper bounded by the sum of the probabilities of all sequences in which the containment sprays worked.

A4.3 Leach Release

If complete melt-through of the containment basemat occurred, then a release of radioactivity to the hydrosphere would generally be possible as a result of leaching of the radioactivity from the core debris by groundwater. In this study, the probability of complete melt-through, and hence the probability of a leach release, is taken to be equal to that estimated in the RSS for meltdown, that is, 4×10^{-5} per reactor year.[†]

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[*] 5×10^{-8} for a "generic" plant.

[**] 2.5×10^{-8} for a "generic" plant.

[†] In this study, the probability of melt-through has been taken to equal the probability of meltdown. However, it may be possible for the debris to be permanently coolable either in the reactor vessel or in the reactor cavity. Permanent in-vessel coolability implies that water is added to the vessel indefinitely; this has not been considered likely for any of the sequences studied. Permanent ex-vessel coolability implies that the core debris forms a rubble bed with fragments of appropriate size, that a sufficient depth of water exists in the cavity, and that some means of replenishing the cavity water exists. For some plant designs, cavity water is guaranteed and replenishment is provided by condensation. In other plants, only limited water would be available in the cavity. Because the question of debris coolability is so plant specific, and because so much phenomenological uncertainty exists, especially with regard to fragmentation, permanent coolability has not been specifically considered in this study. Permanent coolability should be recognized as a possibility, however. The omission of coolability adds to the conservatism of the present study.

Whereas the other two types of releases to liquid pathways would occur over a relatively short period of time, this release could take place over a comparatively extended time period. As soon as groundwater contacted the melt, it would begin to leach out the various nuclides. Initially there could be a delay of many years before groundwater contacted the melt. However, it is more likely that some groundwater contact would occur in most situations within a year after melt-through.

If groundwater were not present at the elevation of the melt, then sufficient cooling of the melt so that water could contact it could take much longer. In such cases the only water available for leaching would be water percolating downward through the unsaturated portion of the soil. Leaching would be expected to proceed at a negligible rate in such situations.

At first, the groundwater might contact just the outer edges of the melt. This could happen at some time before the melt had completely solidified. Groundwater contact with the bulk of the melt, however, could not take place before complete solidification had occurred. Consequently, leaching of the periphery might be initiated somewhat earlier than leaching of the bulk of the melt.

The surface area of the melt debris available for leaching would depend partly upon the composition of that debris. It has been found that interaction of simulated melts with limestone concrete generally produces spongy porous masses.[17] In contrast, simulated melts interacting with siliceous concretes produce glassy masses which are generally porous only on the outside but which are often cracked and fractured throughout.[17,*] Interaction of an actual melt with groundwater would be likely to cause fracturing beyond that observed for the simulated melts. As a result, the effective surface areas of limestone-based melts would probably be orders of magnitude greater than the apparent exterior surfaces. Even siliceous melts would probably have effective areas at least several times greater than the apparent exterior surfaces. As a result, the rate of leaching of any melt would probably not be limited by the exterior surface area of the melt.

The possibility exists that the heat generated during the melt-through might result in the creation of a rather impervious layer around the melt. Such a layer might retard contact of the groundwater with the melt for many years by effectively sealing the melt from the surrounding region. This isolation of the melt would be possible only in certain types of soils and concretes and only in certain melt-through scenarios. It is not thought to be too likely; consequently, this possibility is not addressed further in this report.

When the groundwater initially contacted the melt, the melt would still be very hot. Inasmuch as leach rates can increase dramatically with increasing temperatures, the initial leach rates might be very large compared to the long-term rates due to such thermal effects. Furthermore, even in some leaching experiments at constant temperature, an initial large pulse of material is leached quite rapidly. This effect might also increase the initial leach rates.

Leach rates are a strong function of the composition of the substance being leached. As a result, the absolute values of the leaching rates for a melt would depend upon the final chemical composition of that melt. As mentioned earlier, the melt would be composed predominantly of concrete and soil residua. Therefore, the overall chemical character of the melt would be determined primarily by the compositions of the concrete and the soil beneath the containment.

Leach rates for a given element can vary at least several orders of magnitude as the overall composition of the substance being leached is changed. For example, any element is generally leached several orders of magnitude more quickly from a calcine than from an annealed glass. Furthermore, addition of any alkali to a glass tends to decrease its leach resistance. Consequently, melts high in alkali content might be expected to leach relatively rapidly; melts high in silica content might be expected to leach somewhat more slowly. At the least, there would probably be a rather wide range of leach rates possible for any given element due to the essentially infinite variety of possible compositions of the final melt.

Most of the currently available data for leach rates is for material produced in carefully controlled laboratory conditions. Furthermore, the materials used for such studies have been manufactured with the goal of retarding the leaching of at least certain constituents of those materials. However, the melt would not be formed under such conditions. Therefore, the leach rates obtained from such studies are not very useful for determining reasonable approximations to core-melt leach rates. They can be useful, at most, for estimating lower bounds to the leach rates.

To determine appropriate leaching parameters for this study, many experimentally-determined leach rates were considered.[27-30] These rates were distilled to obtain two sets of information: approximate relative leach rates for different elements; and, appropriate ranges for the absolute rate for each element.

The resulting relative rates are presented in Table A12. As can be seen in that table, in general, elements of low valence and/or small ionic radius tend to be leached most quickly from any substance. For example, of the ions of potential concern after a melt-through, Cs^+ and I^- are among the most leachable and Ce^{+3} and Pm^{+3} are among the least leachable.

The effect of varying the leach rates was taken into account in the calculations in this study by considering five different sets of leach rates. For each set, the relative rates presented in Table A12 were used. The five sets differed in absolute rates only, with the most quickly leached element in each set being leached either instantaneously, or in 10^{-3} years, in 10^{-1} years, in 10^0 years, or in 10^1 years. The five sets were chosen to generously bracket the realm of possibility.

Table A12. Relative Leach Rates

Elements	Relative Leach Rates
I	10^0
Mo,Cs,Sn,Sb,Te	10^{-1}
Co,Sr,Ba,Ru	10^{-2}
Ni,Ag,Ce,Nd,Pm,Sm, Eu,Np,Pu,Am,Cm	10^{-3}

In determining the appropriate ranges of leach rates, no consideration was given to solubility limitations. It was assumed that all the water would be available for leaching which would be necessary to attain maximum removal rates. In practice, in regions of low groundwater flow rates, the amount of certain materials removed from the vicinity of the melt might be restricted by solubility considerations. That is, the presence of certain ions in solution might retard the entry of either more of these ions or else other ions into solution. In regions of relatively high groundwater flow rates, this would most likely not be a restriction for the most important radionuclides due to the comparatively rapid turnover of the water responsible for the leaching.

A5 Release Fractions

The release of radionuclides from the melt into the containment atmosphere would occur more or less continuously over the period of the meltdown. The particular nuclides and the amounts of them released would depend critically upon the specific accident sequence which were followed. In the RSS, the radionuclides present in the melt were grouped into 7 classes, according to their expected physical behaviors during meltdown. The fraction of any element which would go into each type of release was taken to be largely a function of the volatility of the chemical form of that element. For each class of nuclide, both the fraction volatilized and the fraction eventually released to the atmosphere outside the containment (the airborne fraction) were approximated in the RSS.

In this study, the magnitudes of the source terms for the various types of releases were based primarily on the tables for the release fractions given in the RSS.[12,31] Additional calculations[26] were performed for this study to determine the airborne release fractions for both PWR-6 and PWR-7 accident categories as a function of melt-through time. All the volatilized radionuclides were assumed to be found either in the airborne release or in the sumpwater release. That is, processes which would tend to remove radionuclides from the containment atmosphere were considered to be unimportant except for those which would result in the nuclides entering the sumpwater.

For those accident categories not involving steam explosions, the release fraction for each of the species volatilized within the containment was taken to be the sum of the appropriate gap, meltdown and vaporization fractions given in the RSS.[12] The leach fraction of each was taken to be the unit's complement of the volatilized release fraction. For PWR categories, the sumpwater fraction was taken to be the volatilized fraction minus the airborne fraction; for BWR categories, the sumpwater fraction was taken to be 0. The depressurization fraction was taken to be the airborne fraction times the appropriate factor to account for the fact that such release occurs only until the containment pressure and the soil pressure are equalized. The resultant release fractions for most accident categories are summarized in Tables A11 and A13-A15.

For those categories which do involve steam explosions, the release fraction for each of the species volatilized was taken to be the sum of the gaseous release fractions given above plus the oxidation release fraction from the RSS.[12] In applying the oxidation release fraction, the entire core was assumed to be involved in the explosions. Thus, for all PWR categories, the maximal amount possible was released to the sumpwater rather than being allowed to stay in the leach source term. The other fractions were then calculated by the prescriptions given above for an accident sequence not involving a steam explosion. The resultant release fractions are given in Tables A11 and A13-A15.

Table A13. Airborne Release Fractions[28]

Release Category	Xe-Kr	I[a]	Cs-Rb	Te-Sb	Ba-Sr	Ru[b]	La[c]
PWR 1	0.9	0.7	0.4	0.4	0.05	0.4	3×10^{-3}
PWR 2	0.9	0.7	0.5	0.3	0.06	0.02	4×10^{-3}
PWR 3	0.8	0.2	0.2	0.3	0.02	0.03	3×10^{-3}
PWR 4	0.6	0.09	0.04	0.03	5×10^{-3}	3×10^{-3}	4×10^{-4}
PWR 5	0.3	0.03	9×10^{-3}	5×10^{-3}	1×10^{-3}	6×10^{-4}	7×10^{-5}
PWR 6	0.3	8×10^{-4}	8×10^{-4}	1×10^{-3}	9×10^{-5}	7×10^{-5}	1×10^{-5}
PWR 7	6×10^{-3}	2×10^{-5}	1×10^{-5}	2×10^{-5}	1×10^{-6}	1×10^{-6}	2×10^{-7}
BWR 1	1.0	0.40	0.40	0.70	0.05	0.5	5×10^{-3}
BWR 2	1.0	0.90	0.50	0.30	0.10	0.03	4×10^{-3}
BWR 3	1.0	0.10	0.10	0.0	0.01	0.02	4×10^{-3}
BWR 4	0.6	8×10^{-4}	5×10^{-3}	4×10^{-3}	6×10^{-4}	6×10^{-4}	1×10^{-4}

[a] Organic iodine is combined with elemental iodines in the calculations. Any error is negligible since its release fraction is relatively small for all large release categories.

[b] Includes Ru, Rh, Co, Mo, Tc.

[c] Includes Y, La, Zr, Nb, Ce, Pr, Nd, Np, Pu, Am, Cm.

Table A14. Leach Release Fractions

	PWR 2-7; BWR 3-4			PWR 1			BWR 1			BWR 2		
	Min	Best	Max	Min	Best	Max	Min	Best	Max	Min	Best	Max
Xe	0	0	0	0	0	0	0	0	0	0	0	0
I	0	0	.010	0	0	.010	0	0	.010	0	0	.010
Cs	0	0	.050	0	0	.050	0	0	.050	0	0	.050
Te	0	0	.010	0	0	.010	0	0	.010	0	0	.010
Sr	.760	.890	.980	.76	.890	.950	.76	.890	.950	.760	.890	.900
Ru	.675	.920	.980	.60	.600	.600	.50	.500	.500	.675	.920	.970
La	.940	.987	.998	.94	.987	.997	.94	.987	.995	.940	.987	.996

Table A15. Sumpwater Release Fractions for Some Release Categories

	PWR 1			PWR 3			PWR 5			PWR 7		
	Min	Best	Max	Min	Best	Max	Min	Best	Max	Min	Best	Max
Xe	0	0	0	0	0	0	0	0	0	0	0	0
I	.29	.30	.30	.79	.80	.800	.96	.97	.970	.990	1.000	1.000
Cs	.55	.60	.60	.75	.80	.800	.94	.99	.990	.950	1.000	1.000
Te	.59	.60	.60	.69	.70	.700	.98	.99	.990	.990	1.000	1.000
Sr	0	.06	.19	0	.09	.220	.02	.11	.240	.020	.110	.240
Ru	0	0	0	0	.05	.295	.02	.08	.325	.020	.080	.325
La	0	.01	.06	0	.01	.060	0	.01	.060	.002	.013	.060

A6 Effect of Plant Design Characteristics

The detailed analyses in this report were based on a single design of a PWR[15] and just one design of a BWR.[20] However, the designs of other nuclear plants differ considerably from those two. No attempt has been made to analyze other designs in detail; however, the salient features that might influence release will be pointed out.

Melt-through times and the final composition of the melt are strong functions of both the type of concrete in the containment floor and the type of soil beneath the site; they are not expected to be significantly influenced by other plant design variables. The influences of concrete and soil types have previously been bounded, and any specific plant is expected to fall between the extremes already considered.

The probabilities for depressurization and sumpwater releases will indeed be strongly influenced by design variables. Melt-through is considered to be equally probable for all plant types; hence, the probability of a leaching release is expected to be very nearly the same for all plants.*

Table A16 describes six plant designs, including the two used here for detailed study. For most plant types, the depressurization and the sumpwater release probabilities cannot be determined without a complete analysis of the specific plant design. Such a detailed analysis of each U.S. nuclear plant design would be far beyond the scope of this study. An attempt has been made, however, to determine qualitatively the effects of design differences between the types listed in Table A16, with respect to the probability of sumpwater release and depressurization release.

Among the parameters that would influence sumpwater release are the following: the thickness and the location of the shield wall; the relative locations of the reactor cavity and the containment sump; the relative elevations of the reactor cavity floor, the containment floor and the sump cavity floor; the diameter of the reactor cavity; the free volume of the containment; and the mechanical strength of the containment building. Any features which tended either to increase the likelihood of shield-wall failure and/or to decrease the probability of containment failure other than by melt-through of the basemat would tend to promote the possibility of sumpwater release.

For any plant design, it can be stated that the probability of a sumpwater release by breaching the shield wall is greatly reduced if the containment sump is located at a higher level than the bottom of the reactor cavity and there is no communication between the two. The reason for this can be seen by examination of Figures A3 and A6. The maximum radius of the melt cavity is below the original bottom of the reactor cavity. If the containment sump floor is higher than the bottom of the reactor cavity - even by a few feet - the remaining thickness of the shield wall is considerably greater and sumpwater breakthrough

Table A16. Plant Design Differences

Design Type	Reference	Remarks
1 - Reference PWR	15	See detailed analysis, this study.
2 - PWR Ice Condenser Plant	32	Low containment volume. Containment sump near reactor cavity - slightly higher elevation.
3 - CE PWR Design	33	Irregularly shaped shield wall. Containment sump at same level as reactor cavity.
4 - SNUPPS PWR	34	Containment floor above reactor cavity.
5 - Reference BWR	20	See detailed analysis, this study.
6 - BWR with containment	35	Secondary containment structure in addition to dry well.

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* The probability of melt-through will be significantly affected by those design variables which relate to the possibility of permanent ex-vessel coolability of the melt. In order to understand the effects of these differences, one would need to know more than is currently known about the fragmentation of core melts and the formation of debris beds, as well as understanding the details of plant construction in detail. For at least some plants, permanent coolability is considered very possible or even highly probable, under almost all accident conditions.[36] There is, however, a contrary body of opinion[37] which holds that even for the most optimistic circumstances permanent coolability has not been sufficiently established. If the core debris is permanently coolable in the reactor cavity, there is little possibility of a release to the subsurface. The present study has taken the conservative course of ignoring the possibility of coolability for all plants.

is much less likely. Conversely, locating the reactor sump near the reactor cavity, but at a lower level, would increase the probability of sumpwater breakthrough.

The probability of depressurization release is affected by all features which relate to the structural integrity of the containment during meltdown. Containments which are designed to withstand relatively high internal pressures are more likely to undergo depressurization into the soil.

A detailed analysis of the Type 1 plant described in Table A16 has already been performed (Figure A2). In this type of plant, the containment floor is at approximately the same level as the floor of the reactor cavity, so that melt-through of the shield wall could lead to sumpwater release to the top of the melt (Figures A6 and A7). The shield wall is regular in shape, except for the incore instrumentation area.

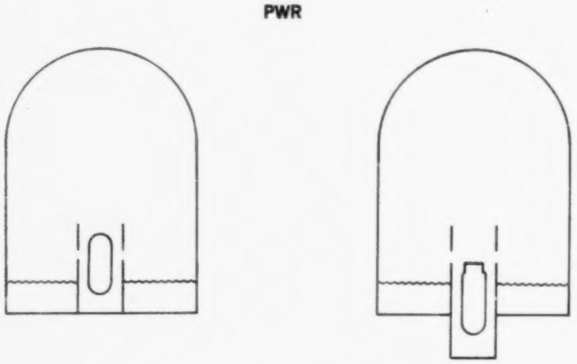

Figure A7. Typical PWR Containment Designs.

Type 2 (PWR) plants[32] have a significantly lower free containment volume than do Type 1 plants. Because of this lower volume, the probability of containment overpressure failure before melt-through is higher. The containment sump is at a higher elevation than the floor of the reactor cavity (Figure A7). These two design features make sumpwater release highly improbable. The lower containment volume also militates against any depressurization release.

Type 3 (PWR) plants[33] have an irregularly-shaped shield wall. The effects of irregularities would have to be determined by a detailed analysis. The containment floor is at the same level as the reactor cavity floor. In the absence of a detailed analysis, it is assumed that the probability of sumpwater and depressurization releases are similar to those for the Reference PWR (Type 1).

Type 4 (PWR) plants[34] are similar to Type 1, except that the floor of the reactor cavity is considerably lower than that of the containment (Figure A7). Although a detailed analysis would be required to determine whether sumpwater release is possible, it can be tentatively assumed that the probability is low. The probability of a depressurization release from such a plant is expected to be approximately the same as that for a Type 1 plant.

Plant Type 5 is the Reference BWR (Figure A3). In this type of plant, failure of the drywell before melt-through is highly probable. The secondary containment building is not designed for high pressure, and therefore wetwell water release and depressurization release are very unlikely to occur.

Type 6 (BWR) plants[35] are similar to Type 5 ones, with the addition of a secondary containment which will withstand relatively high pressures (Figure A8). Detailed analysis would be required to determine whether the containment would survive after melt-through. If survival were possible, then depressurization and water releases could occur. However, the release fractions would be very different from those from Type 1 plants and would have to be analyzed in detail.

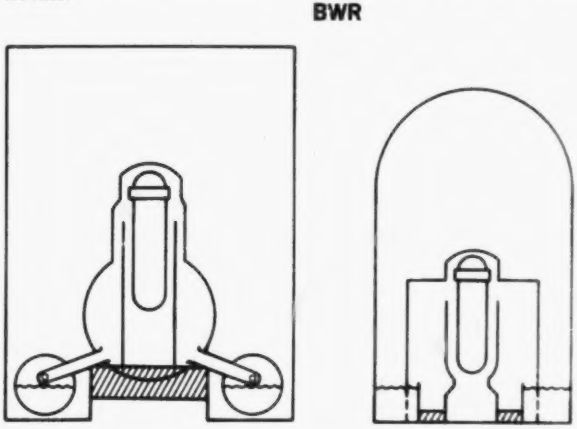

Figure A8. Typical BWR Containment Designs.

A7 General Remarks

The phenomena of a core-melt accident are exceedingly complex and, in many cases, not yet well understood. Melt-through of the containment basemat is especially difficult to predict with precision. Growth and cooling of the core debris mass in the underlying soil is governed by parameters which cannot yet be well defined. Accurate physical models are not currently available. Because of the many uncertainties involved, the predicted physical and chemical forms of the solidified mass should be considered to be estimates. The band of uncertainty surrounding these estimates is believed to be wide, and not currently quantifiable.

The most important parameters influencing the chemical and physical form of the melt would be the type of concrete in the basemat and the composition and water content of the underlying soil. In any core-melt accident, the reactor materials -- fuel, cladding, and structural materials -- would be highly diluted, and would represent only a very small fraction of the total final mass of the melt. If the concrete and subsoil were siliceous, the solidified mass would be largely silica. If the concrete and subsoil were largely calcium carbonate, the mass would consist principally of lime.

The flow of gas through the melt during its growth would probably lead to a high void fraction. As a result, the solidified mass could often be characterized as porous, rather than as compact and glassy; the effective surface area of the mass would be expected to be many times greater than the geometric exterior surface area. Because of the substantial dilution of the denser core constituents with concrete and soil and because of the expected porosity, the specific gravity of the melt would probably be comparable to that of the subsoil.

The probabilities of release cannot be precisely quantified. Principal determinants are the plant design and the accident sequence.

Although the initial inventory of the core can be computed with reasonable accuracy, apportionment of this inventory to the melt debris, the containment atmosphere, and the sumpwater is subject to large uncertainties.

The chemical and physical form of the melt, the probabilities of releases, and the release fractions are inputs to all following steps of the problem. The uncertainties mentioned can be compounded in following steps.

An effort has been made in this analysis to utilize "best estimates" rather than conservative estimates. It is recognized that this procedure is somewhat subjective. Other analysts might consider the estimates of this appendix to be either over- or under-conservative.

Some values are limited by physical law. For example, the volumes of concrete and soil affected are limited by the decomposition and melting enthalpies of the concrete and soil and by the available decay power of the melt. Uncertainties exist in the fraction of soil water evaporated and the upward heat transfer from the melt; these uncertainties are not expected to produce errors in the decomposed volume of greater than a factor of two.

The greatest impact on the uncertainty of the source description in this study will be given by uncertainties of the probability of basemat melt-through, the probability of sumpwater escape, and the fraction of sumpwater escaping. Unless basemat melt-through occurs and/or sumpwater escapes, there will be no hydrospheric consequences whatever. It is believed, on the basis of a number of WECHSL and INTER calculations, that the estimated probability of basemat melt-through is too high; however, sufficient information is not now available to quantify this belief. If shield-wall failure can be ruled out, the probability of sumpwater escape through the melt hole could be reduced essentially to zero for plants without any communication between the sump and reactor cavities. The impact of the other uncertainties cannot be easily addressed but is intuitively believed to be small.

A summary of the "best estimates" and of the estimated uncertainties, is given in Table A17.

Table A17. Uncertainties in Core Melt Source Terms

Phenomenon	Nominal Value	Minimum Value	Maximum Value
Probability of base melt-through (given that meltdown occurs)	1.0	0.0	1.0
Cooling time	2 yrs	0.5 yr	3 yrs
Groundwater contact	1 yr	0.05 yr	3 yrs
Porosity of solidified mass	0.15	0.05	0.25
Probability of solidified mass	0.15	0.05	0.25
Probability of complete depressurization (per reactor year)	6×10^{-7}	0.0	1×10^{-5}
Probability of partial depressurization release (per reactor year)	1×10^{-5}	0.0	1×10^{-5}
Fraction of atmospheric activity released in depressurization	0.6	0.0	1.0
Probability of shield-wall failure	0.5	0.0	1.0
Probability of sumpwater escape (per reactor year)	5×10^{-6}	0.0	1×10^{-5}
Fraction of sumpwater escaping	0.1	0.0	0.5

ADDENDUM A1

Calculations for Cooling of Melt by Groundwater

Models are available for computing water migration under special circumstances, e.g., the numerical models of Bowman[21] and of Keller, et al.,[22] or the analytical model of Morrison.[23] None of these models are closely applicable to the situation at hand, although that of Keller, et al., comes closest. Because a previously defined model was not readily available, the method of Keller, et al., was modified for the calculations.

The relevant assumptions and physical laws in that model are:

1. Darcian flow in the medium (the fluid velocity is dependent on permeability, fluid viscosity, and pressure gradient);
2. conservation of mass;
3. conservation of energy (heat transfer between the fluid and the medium is allowed);
4. equation of state behavior (the approximations of Keller, et al.,[22] have been used to obtain fits to steam table data).

In considering water migration, a porous medium in one-dimensional cylindrical geometry (Figure A1.1) is employed. The inner boundary r_0 is heated by the melt debris, and the outer boundary r_1 is considered far enough away to be unaffected by the heat of the melt debris. The inner boundary is maintained at a pressure P_0 and a temperature T_0 by the melt debris, and the outer boundary is maintained at a pressure P_1 and a temperature T_1. The region between r_0 and r_1 contains a porous, permeable, water-containing medium, e.g., soil or gravel.

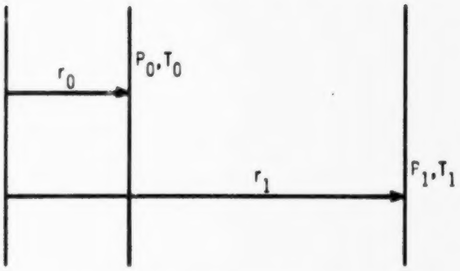

Figure A.1. Geometry Employed for Cooling Calculations

The equations have been solved by the method of central finite differences. The full solution is only carried as far as the first point showing subcooled water. Beyond this point conservation of mass and the Darcy equation are combined to compute pressures. The details of the solution depend on the boundary conditions and properties of the medium. However, two general classes of solution have appeared, depending on whether P_0 is less than, or greater than, P_1. In every case, T_0 was chosen to be 1300°K and T_1 to be 293°K. (The temperature of 1300°K is an estimate of the temperature at which breakdown of gravel commences; the actual melting point is generally higher.) The solution undoubtedly also depends on these temperatures; however, these are typical of those to be found in the INTER calculations.

For the case with $P_0 < P_1$, initial flow is toward the inner boundary, $r = r_0$. However, after a short time, the medium becomes sufficiently hot near the inner boundary to vaporize the fluid. The pressure within the medium then builds up near the inner boundary and the flow splits, with part of the fluid continuing to flow inward and with part flowing outward. Eventually, of course, pressure equilibrium is reestablished and all the flow is inward. However, this simple model does not include ablation of the inner boundary. In the actual melt situation, the inner boundary is moving outward (at least initially) so that the divided flow should continue.

For the case with $P_0 \geq P_1$, the flow is uniformly outward but at a very slow rate. Again, the movement of the inner boundary would alter the situation. Water vapor trapped in the pore spaces would be incorporated into the melt.

References

[1] M. J. Bell, *ORIGEN, the ORNL Isotope Generation and Depletion Code*, ORNL-4628, Oak Ridge National Laboratory, 1973.

[2] ANS Standards Committee, Proposed ANS Standard 5.1, "Decay Heat Power in Light Water Reactors," American Nuclear Society, June 1978.

[3] D. E. Bennett, *SANDIA-ORIGEN Users Manual*, SAND79-0299, Sandia Laboratories, Albuquerque, NM, to be published.

[4] W. B. Murfin, *A Preliminary Model for Core/Concrete Interactions*, SAND77-0270, Sandia Laboratories, 1977.

[5] H. Alsmeyer, W. Murfin, M. Reimann, "Hydrodynamische und thermische Modelle zur Wechselwirkung einer Kernschmelzen mit Beton," *PNS Halbjahresbericht 1978/1*, KfK2700, Kernforschungszentrum Karlsruhe, GmbH, September 1978.

[6] D. A. Powers and A. W. Frazer, *VISRHO - A Computer Program for Estimating the Viscosity and Density of Complex Silica Melts*, SAND76-0649, Sandia Laboratories, Albuquerque, NM, 1977.

[7] P. Misaelides and G. Ondracek, *Abschätzung der Liquidus und Soliduskurven von Beton-Corium-Schmelzen*, PNS-Bericht 218/78 (Primärbericht), Kernforschungszentrum Karlsruhe, GmbH, March 1978.

[8] M. Peehs and K. Mollwitz, "Untersuchung der metallurgischen und chemischen Wechselwirkung zwischen Kernschmelzen und RDB-Wand," Abschlussbericht RS74a, Kraftwerk-Union, May 1976.

[9] U.S. Nuclear Regulatory Commission, *Reactor Safety Study* (WASH-1400), NUREG-75/014, NRC, Appendix V, 1975.

[10] U.S. Nuclear Regulatory Commission, *Reactor Safety Study, (WASH-1400)*, NUREG-75/014, NRC, Appendix VIII, 1975.

[11] J. Fischer, J. D. Schilb, and M. G. Chasanov, "Investigation of the Distribution of Fission Products Among Molten Fuel and Reactor Phases. I. The distribution of fission products between molten iron and uranium dioxide," *J. Nuc. Mat., 48*: 233-240, 1973.

[12] U.S. Nuclear Regulatory Commission, *Reactor Safety Study* (WASH-1400), NUREG-75/014, NRC, Appendix VII, 1975.

[13] ANS Standards Committee, Proposed ANS Standard, "Decay Energy Release Rates," American Nuclear Society, 1973.

[14] B. I. Spinrad, "Evaluation of Fission Product Afterheat," NUREG-0018-2, Oregon State University, 1976.

[15] Virginia Electric and Power Co., *Surry Power Station Units 1 and 2, Final Safety Analysis Report*, VEPCO, 1969.

[16] H. S. Levine, "Non-condensible Gas Evolution," from *Core-Meltdown Experimental Review*, SAND74-0382 (NUREG-0205 Rev.), Sandia Laboratories, 1977.

[17] D. A. Powers, Sandia Laboratories, Albuquerque, NM, personal communication.

[18] K. Shure, *Fission Product Decay Energy*, WAPD-BT-24, Bettis Atomic Power Laboratory, 1961.

[19] D. A. Power, *Molten Core Concrete Interaction Project Description*, 189 No. A-1019, Sandia Laboratories, 1975.

[20] Philadelphia Electric Co., *Peach Bottom Nuclear Plant Final Safety Analysis Report*, Philadelphia Electric Co., 1970.

[21] B. R. Bowman, *TWIG, A Computer Code for the Calculation of Two-Dimensional Ideal Gas Flow in Porous Media*, UCID-16320, Lawrence Livermore Laboratory, 1972.

[22] C. E. Keller, A. H. Davis, and J. N. Steward, Jr., *The Calculations of Steam Flow and Hydraulic Fracturing in a Porous Medium with the KRAK Code*, UC-34, Los Alamos Scientific Laboratory, 1974.

[23] F. A. Morrison, Jr., *Transient Multiphase Multicomponent Flow in Porous Media*, UCRL-74327, Lawrence Livermore Laboratory, 1972.

[24] R. L. Ritzman, et al., "Release of Radioactivity in Reactor Accidents," in *Reactor Safety Study* (WASH-1400), NUREG-75/014, NRC, Appendix VII, 1975.

[25] B. D. Zak and A. J. Russo, "Fission Product Transport and Removal within Containment," Chapter 8 of *Core-Meltdown Experimental Review*, SAND74-0382 (NUREG-0205 Rev.), Sandia Laboratories, 1977.

[26] R. S. Denning, Battelle Columbus Laboratories, personal communication, 1978.

[27] K. Wolfsberg, "Sorption-Desorption Studies of Nevada Test Site Alluvium and Leaching Studies of Nuclear Test Debris," LA-7216-MS (UC-11), LASL, 1978.

[28] J. G. Moore, H. W. Godbee and A. H. Kibbey, "Leach Behavior of Hydrofracture Grout Incorporating Radioactive Wastes," Nuclear Technology 32, 39-52, 1977.

[29] J. E. Mendel, "A Review of Leaching Test Methods and the Leachability of Various Solid Media Containing Radioactive Wastes," BNWL-1765, BNWL, 1973.

[30] U.S. Nuclear Regulatory Commission, *Liquid Pathway Generic Study*, NUREG-0440, NRC, Appendix A, 1978.

[31] U.S. Nuclear Regulatory Commission, *Reactor Safety Study* (WASH-1400), NUREG-75-014, NRC, Appendix VI, 1975.

[32] Tennessee Valley Authority, *Watts Bar Nuclear Plant, Preliminary Safety Analysis Report*, TVA.

[33] Baltimore Gas and Electric Co., *Calvert Cliffs Nuclear Power Plant Units 1 and 2, Final Safety Analysis Report*, Baltimore Gas and Electric Co., 1971.

[34] Kansas City Power and Light Co., *Standardized Nuclear Unit Power Plant System, Preliminary Safety Analysis Report*, Kansas City Power and Light Co.

[35] Mississippi Power and Light Co., *Grand Gulf Nuclear Station Units 1 and 2, Preliminary Safety Analysis Report*, Mississippi Power and Light Co.

[36] D. Walker, et al., "Indian Point and Zion Near Site Study," Commonwealth Edison Company, Consolidated Edison Company, Power Authority of the State of New York, 1980.

[37] W. B. Murfin, "Summary of the Zion/Indian Point Study," SAND80-0617, NUREG/CR-1409, Sandia National Laboratories, 1980.

APPENDIX B

Groundwater and Surface Water Transport

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APPENDIX B

Groundwater and Surface Water Transport

Authors: S. J. Niemczyk
K. G. Adams
W. B. Murfin (River System Model)

B1 Introduction

The modeling of the movement of radionuclides through the hydrosphere is discussed in this appendix. In particular, the models used to consider groundwater and surface-water transport are presented. In addition, the modeling of sediment movements and of soil-nuclide and sediment-nuclide interactions is addressed.

A general discussion of the salient characteristics of transport through the hydrosphere is presented in Chapter 3. The models used to describe that transport in this study are presented in this appendix. The groundwater transport models are described in Section B2 and the surface-water transport models are considered in Section B3. In the last section of this appendix, B4, the results of sensitivity studies for all of the transport models except for the river model are summarized; the results of less extensive parametric studies for the river model are also presented in that section.

In this study, five distinct types of generic sites have been defined on the basis of differing surface hydrologies. These five are the following: "dry," lake, estuary, ocean, and river. Transport is considered separately at each of the five types of sites.

B1.1 Hydrospheric Transport and Dispersion

The general movement of a radionuclide in any waterbody can be described by

$$\frac{dc}{dt} = u_x \frac{dc}{dx} - u_y \frac{dc}{dy} - u_z \frac{dc}{dz} + D_x \frac{d^2c}{dx^2} + D_y \frac{d^2c}{dy^2} + D_z \frac{d^2c}{dz^2} - \lambda c + S(t) , \qquad (B1\text{-}1)$$

where c is the concentration of the radionuclide in the water; u_x, u_y, and u_z are the x, y and z components of the velocity, respectively; D_x, D_y and D_z are the dispersion coefficients for the x, y and z directions, respectively; λ is the radioactive decay constant of the material; and S(t) is the rate of the release of the material into the water. The first three terms in this equation describe the transport of the radionuclide due to major convective and advective movements of the water itself. The next three terms describe any additional dispersal of the radionuclide due to other factors.

Several processes tend to cause dispersal of the material beyond that caused by the basic movements of the water. During transport of radionuclides through the ground, at least four major phenomena contribute to such additional dispersal: molecular diffusion, microscopic dispersion, geohydrological dispersion, and physicochemical interactions between the soil and the nuclides. For most groundwater transport, the geohydrological dispersion completely dominates the diffusion and the microscopic dispersion. Consequently, the total physical dispersion coefficients are essentially just the geohydrological ones. The various physicochemical interactions occurring between the nuclides and the soil substrate cause substantial additional spreading of the nuclide mass as it moves through the ground.

In a similar fashion, at least three phenomena contribute to additional dispersal of materials entering surface waterbodies: diffusion, hydrological (macroscopic) dispersion and interactions between the sediment and the nuclides. There, as in the groundwater case, the macroscopic dispersion normally dominates the diffusion. Consequently, the appropriate physical dispersion coefficients are essentially just the hydrological ones. In certain waterbodies, sediment-nuclide interactions can cause noticeable additional dispersal to take place.

B1.2 Sediment and Soil Effects

B1.2.1 Sediment and Soil Behavior

Interactions occurring both between the soil and any radionuclides present in groundwater and between any sediments and the radionuclides in surface water can remove some of the radionuclides from solution. As a result, such interactions tend to lower the concentrations of the radionuclides in solution. Furthermore, such interactions tend to significantly retard the movement of certain radionuclides relative to the movement of the transporting water. The same types of basic interactions take place with both soil and sediment particles.

In the ground, any soil particles in contact with water containing radioactivity can adsorb radionuclides from the groundwater. After the nuclides are initially attached to soil in a given location, they remain in those positions until most of the nuclide mass in the water has passed. That is, they remain until the conditions favor detachment from the soil substrate. Then they reenter the water and subsequently reattach themselves to other soil particles further downgradient. In many respects, the movement of radionuclides through the soil is analogous to the movement of radionuclides through an exchange column.

In a surface waterbody, any sediment either in suspension or else in other contact with the water can adsorb radionuclides from the water column. After the radionuclides are attached to the sediments, they can be removed from the water by sedimentation. After such deposition, substantial quantities of sediment can be resuspended by local water movements. These sediments can either be redeposited on the waterbody floor or can be transported by movement of the water and subsequently be deposited in new locations.

If sediment accumulation is relatively rapid, contaminated sediment can be buried and essentially lost to further contact with water. Only resuspension, diffusion and bioturbation (turnover of the sediments by biological organisms) are expected to counteract the effects of such burial. For rapid enough deposition, only biological turnover may be significant. If such burial does not occur and if later conditions favor detachment of the radionuclides from the sediment, then the radionuclides can reenter the water and can eventually be adsorbed to other sediment particles.

Radionuclides removed from the water by interaction with sediment particles and subsequent deposition of that sediment can reenter the water column by diffusional processes. However, those processes which favor adsorption and other such interactions tend to work against the diffusional processes. Although there can be a tendency for diffusion back into the water column, subsequent adsorption can occur during the transit through the interstitial waters of the sediment particles. Consequently, the diffusing radionuclide cannot always reach the water column before decay.

B1.2.2 Differences Between Soil and Sediment

The differences in the effects of the interactions in soil and in sediment are due to two factors: the soil is fixed in space whereas the sediment is relatively free to move; and the proportion of soil to water is much larger than the ratio of sediment to water. The differences in the relative mobilities of soil and sediment result in several distinctions between the effects caused by interaction of nuclides with them. First, certain reactions which can cause relative retardation of a nuclide's movement in the soil might not as effectively cause such a delay in the sediments. For example, precipitation might not be as effective in hindering movement if it occurs in a surface waterbody as if it occurs in the soil. A precipitate residing in "loose" sediment can often be moved very readily by surface water advection and convection, whereas in the soil it might have its movement greatly retarded by the soil's pore structure.

A second difference caused by the mobilities is that burial of contaminated sediments can occur. Such "burial" cannot occur in soil unless the groundwater flow is massively disrupted.

A third distinction which results from differing mobilities is that contaminated sediments can be resuspended. Consequently, they can be exposed to much more water than soil particles can be. This resuspension has at least two effects. First, it permits the nuclides in solution to be exposed more completely to the sediments and thus interactions between the two are promoted. In particular, attachment of large quantities of the nuclides to the resuspended sediments and eventual removal from the water can transpire very rapidly. Second, the resuspended sediments can be moved substantial distances. Thus resuspension can permit significant quantities of adsorbed radioactivity to be moved to remote regions in short periods of time. There is no phenomenon in the soil comparable to resuspension. As a result, a large portion of a radionuclide mass can be confined to a relatively limited region of soil whereas it might not be as confined in sediments over long periods of time.

An additional difference caused by the mobility of the sediment particles is that the sources of many sediments can be far from the locations in which they are found. Consequently, the composition of sediment in a region can be completely different from that of nearby soils. It can also be much more diverse than that of the soil if several sources contribute to the sediment pool.

The primary effect of the larger ratio of soil to water than sediment to water is that much more of the total mass of a nuclide can reside on the soil than on the sediment. Thus the soil can be substantially much more effective in lowering the radionuclide concentration in the water than the sediment can be. Furthermore, the soil can cause much larger retardation of the movement of any radionuclide mass then the sediment can.

B1.2.3 Soil-Nuclide and Sediment-Nuclide Interactions

There are several different significant interactions which can occur between either soil or sediment particles and the nuclides in solution. The tendencies of radionuclides to undergo any of these interactions are determined primarily by both the state of the water and the thermodynamic properties of the individual radionuclides. The effects of these factors is discussed in Subsection B.2.2.3 and in the first addendum of this appendix, Addendum B1.

The amount of interaction between the soil or the sediment and any dissolved nuclides is a strong function not only of the character of the water but also of the composition of the soil or the sediments. Sediment particles with large numbers of reaction sites tend to undergo more interactions.*

The amount of interaction between the soil or sediments and any nuclides is also dependent upon the size of the individual particles. Smaller particles have proportionally larger surface areas and consequently expose relatively more sites at which reactions can occur.

B2 Groundwater Transport

Initially, any release to the hydrosphere after a meltdown is usually expected to be to the substratum beneath the containment.

B2.1 Models

If exposure is directly from contaminated aquifer water, then the concentration of the radioactivity at the point of withdrawal from the aquifer is important. In contrast, if exposure is from subsequently contaminated surface water, then the amount of radioactivity entering the surface waterbody from the aquifer is relevant. Models for calculating the concentration at any point within the aquifer and for calculating the amount of material crossing the aquifer-waterbody interface are presented in this section.

Both models involve certain simplying assumptions that permit analytical solutions to be obtained for some types of releases from sources of certain shapes. The analytical solutions have been combined with convolution techniques as necessary to account for complex release functions. Both models are essentially the same as those used in the NRC Liquid Pathway Generic Study (NUREG-0440).[1]

B2.1.1 Concentration Model

The equation which describes the transport and dispersion of a radionuclide by groundwater through an aquifer or other substratum can be written as

$$a\frac{dc}{dt} = - u_x \frac{dc}{dx} - u_y \frac{dc}{dy} - u_z \frac{dc}{dz} + D_x \frac{d^2c}{dx^2} + D_y \frac{d^2c}{dy^2} + D_z \frac{d^2c}{dz^2} - \lambda ac + a S(t) \quad ,\text{(B2-1)}$$

where a is the retention factor of the radionuclide for the substratum under consideration; c is the average concentration of the radionuclide in the water and the substratum; and the other parameters are as defined previously (Subsection B1.1). The retention factor a is given by

$$a = 1 + (\rho_b/\epsilon)K_d, \qquad \text{(B2-2)}$$

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*For example, particles composed of quartz-derived sand have relatively few sites at which interactions can occur; consequently, few reactions take place between clean sand and most radionuclides. In contrast, particles composed of clay have numerous sites at which reactions can take place; therefore substantially more interactions occur between clay and many radionuclides than occur between sand and those nuclides. As a result, a much higher concentration of nuclides is expected to reside in the clay sediment than in clean sand sediment under the same conditions.

where ρ_b is the bulk density of the substrain; ϵ is the porosity of that layer; and K_d is the distribution coefficient for the radionuclide.

K_d is generally assumed to be the equilibrium adsorption constant for the nuclide in the medium. Actually, it is a measure of the net retardation of the nuclide caused by any of several potentially important physicochemical processes; adsorption is just one of these processes. (Furthermore, K_d may not be the equilibrium adsorption constant because equilibrium conditions may not exist.)

K_d is given by

$$K_d = \frac{\text{mass of radionuclide adsorbed per gram of adsorbent (soil)}}{\text{mass of radionuclide dissolved per milliliter of solvent (water)}} \quad . \tag{B2-3}$$

The ratio of the total amount of a given nuclide adsorbed on the soil to the total amount in solution is given by $(\rho_b/\epsilon) \cdot K_d$. A typical value for (ρ_b/ϵ) is 5; consequently, for most values of K_d, the ratio of radioactivity on the soil to that in solution is approximately 5 times the value of K_d. K_d typically ranges from 0 to 10^5.

Equation (B2-1), the equation which describes the transport of a radionuclide, is obtained by combining the equation for conservation of mass under flow conditions with that for flow under a hydraulic gradient. The equation for flow under such a gradient is given by Darcy's Law, that is,

$$u_x = -K \frac{d\phi}{dx} \; , \tag{B2-4}$$

where u_x is the magnitude of the groundwater velocity in the x direction; K is a proportionality constant known as the permeability; and $(d\phi/dx)$ represents the hydraulic gradient in the x direction.

u_x is actually a fictitious speed and represents the apparent speed of the water; it would be the magnitude of the velocity of the water if it were flowing across the entire interface considered. The actual velocity of the water is obtained by taking into account the fact that the water flows only through the medium's pores. The actual groundwater velocity is given by

$$u_x = -\frac{K}{\epsilon_e} \frac{d\phi}{dx} = u_x \, (\epsilon_e)^{-1} \; . \tag{B2-5}$$

If the average speed of the groundwater is given by u_x, then the average speed of a radionuclide is given by $(u_x) \cdot (a)^{-1}$. As an example of the amount of retardation, if K_d were 100 and (ρ_b/ϵ) were 5, it would take approximately 500 times as long for the nuclide mass to move the same distance as the water.

Geological formations are typically rather irregular. However, the detailed information which would be required to model the inhomogeneities of any given geological site is not generally available. Consequently, for this study, the substrate is assumed to be a homogeneous hydrogeological one in which the permeability, the porosity, the effective porosity, the hydraulic gradient, the retention factors and the dispersion coefficients are all assumed to be independent of location; that is, they are all taken to be constants. Furthermore, the physicochemical interactions between the soil and the radionuclides in solution are taken to be adequately described by equilibrium conditions.

Under these conditions, the solution of Equation (B2-1) is given by

$$c_i = (\epsilon_e a_i)^{-1} \; Q_i \, c_{ix}(x,t) \, c_{iy}(y,t) \, c_{iz}(z,t) \, \Lambda_i(t) \; , \tag{B2-6}$$

where c_i is the concentration of radionuclide i in solution at point (x,y,z) at time t; ϵ_e is the effective porosity; a_i is the retention factor for i; Q_i is the initial quantity of nuclide i at the source; c_{ix}, c_{iy} and c_{iz} are the solutions for the x,y and z coordinates, respectively; and Λ_i describes decay. The product $c_{ix} \cdot c_{iy} \cdot c_{iz}$ represents the concentration of nuclide i if all space is occupied by water; $(\epsilon_e)^{-1}$ takes into account that only a fraction ϵ_e of space is occupied by water and $(a_i)^{-1}$ adjusts for the fact that only a fraction of nuclide i is in the water.

For an instantaneous release from a point source at the origin at $t = 0$, the concentration can be written as

$$c_i = (\epsilon_e a_i)^{-1} \; Q_i \, c_{ixI} \; c_{iyI} \; c_{izI} \; \Lambda_i \; , \tag{B2-7}$$

where

$$c_{ixI} = (4\pi D_x t/a_i)^{-1} \; \exp\{-[(x-ut/a_i)^2/(4D_x t/a_i)]\} \; ;$$

$$c_{iyI} = (4\pi D_y t/a_i)^{-1} \; \exp\{-[y^2/(4D_y t/a_i)]\} \; ;$$

$$c_{izI} = (4\pi D_z t/a_i)^{-1} \; \exp\{-[z^2/(4D_z t/a_i)]\} \; ;$$

$$\Lambda_i = \exp(-\lambda_i t) \; . \tag{B2-8}$$

In general, solutions for more realistic noninstantaneous releases are not obtained quite as easily as those for idealized instantaneous ones. However, for an arbitrary continuous release of radioactivity, the solution of Equation (B2-1) can be written as

$$f(t) = \int_0^t S(t') \, f'(t-t') \, dt' \; , \tag{B2-9}$$

where f(t) is the solution at time t for the continuous release; S(t′) is the rate of continuous release by the source at time t′; and f′(t-t′) is the solution at time (t-t′) for an instantaneous release. Thus, one can straightforwardly obtain concentrations for continuous releases from point sources by combining Equation (B2-9) with solutions like that of Equation (B2-7).

For example, for a typical leaching release of the form be^{-bt} from a point source, the convolution of the expression for the concentration for an instantaneous release can be written as

$$c(t) = \int_0^t be^{-bt'}\, c_{|x|}(t-t')\, c_{|y|}(t-t')\, c_{|z|}(t-t')\, dt'$$

$$= (n_e a)^{-1}\ (4D_x/a)^{-1/2}\, b\ \exp(-\beta)\,(2\alpha^{1/2})^{-1}$$

$$\cdot\ \{\exp[2(\alpha\gamma)^{1/2}]\ \mathrm{erfc}[(\gamma/t)^{1/2}+(\alpha t)^{1/2}]$$

$$+ \exp[-2(\alpha\gamma)^{1/2}]\,\mathrm{erfc}[-(\gamma t)^{1/2}+(\alpha t)^{1/2}]$$

$$- 2\exp[-2(\alpha\gamma)^{1/2}]\}\ , \qquad \text{(B2-10)}$$

where

$$\alpha \equiv (u^2/4D_x a) + \lambda - b;$$

$$\beta \equiv -xu/(2D_x) + bt; \qquad \text{(B2-11)}$$

$$\gamma \equiv x^2 a/(4D_x);$$

and erfc denotes the unit's complement of the error function and is defined as[2]

$$\mathrm{erfc}(z) \equiv (2/\pi^{1/2}) \int_z^{\infty} \exp(-t^2)dt = 1-\mathrm{erf}(z)\,. \qquad \text{(B2-12)}$$

Likewise, for a constant rate of release b from a point source, the convolution integration produces the same result as that given in Equation (B2-10), except that α and β must be redefined such that

$$\alpha \equiv (u^2/4D_x a) + \lambda,$$

$$\beta \equiv -xu/(2D_x). \qquad \text{(B2-13)}$$

Unlike the exponential release, the source for a constant release vanishes after a time $(b)^{-1}$; therefore, this time forms an effective upper limit to the integration period appropriate for a constant rate of release.

B2.1.2 Flux Model

The amount of a radionuclide reaching any surface waterbody is given by the flux across the aquifer-waterbody interface as a function of time. The flux of material passing across a plane (yz) perpendicular to the direction of flow (x) can be written as

$$F_x = Q\ \epsilon_e \iiint (u_x \frac{dc}{dx} - D_x \frac{d^2c}{dx^2})\ dx\,dy\,dz\ , \qquad \text{(B2-14)}$$

$$F_x = Q\ \epsilon_e \iint (u_x c - D_x \frac{dc}{dx})\ dy\,dz\ . \qquad \text{(B2-15)}$$

Thus, for an instantaneous release from a point source into an aquifer or other subsurface waterbody, the flux across an interface at a distance x from the source is

$$F_{x1} = Q\ [x + (ut/a_1)]\ \cdot\ [2t(4\pi D_x t/a_1)^{1/2}]^{-1}$$

$$\exp\{-[(x-ut/a_1)^2/(4D_x t/a_1)] - \lambda_1 t\}\ . \qquad \text{(B2-16)}$$

For a continuous release of radioactivity from the source, Equation (B2-9) can be used, with f′(t-t′) being the flux for an instantaneous release from a source of the same shape. For example, for a release of the form be^{-bt} from a point source into an aquifer or other subsurface waterbody, the convolution integral of the flux produces

$$F_x(t) = \int_t^{t'} be^{-bt'} f_{x1}(t-t)\ dt$$

$$= QB\,[4(4D_x/a)\,]\exp(-\beta)$$

$$\cdot \Big\{x\gamma\ \big[\exp\{2(a\gamma)\}\mathrm{erfc}\{(\gamma/t) + (\alpha t)\}$$

$$+ \exp\{-2(\alpha\gamma)\}\mathrm{erfc}\{(\gamma/t) -(\alpha t)\} - 2\exp\{-2(\alpha\gamma)\}\big]$$

$$+ ua\,\alpha\ \big[\exp\{2(\alpha\gamma)\}\mathrm{erfc}\{(\gamma/t) + (\alpha t)\}$$

$$+ \exp\{-2(\alpha\gamma)\}\mathrm{erfc}\{(\gamma/t) -(\alpha t)\} - 2\exp\{-2(\alpha\gamma)\}\big]\Big\}$$

$$\text{(B2-17)}$$

where α, β, and γ are as they were defined in Equations (B2-11).

Similarly, for a constant rate of release b from a point source, the convolution integral of the flux produces the same result as Equation (B2-17), except that α and β must be redefined as they are in Equation (B2-13).

B2.2 Parameters

B2.2.1 Site Characteristics

B2.2.1.1 Compilation

A survey of all the existing and many proposed LWR (light water reactor) sites was conducted to determine the variations of the hydrological parameters which actually exist at reactor sites.[3] That is, an attempt was made to characterize a representative substratum for an LWR site. The information obtained included geologic stratigraphies for 50 feet below the containments, permeabilities, porosities, hydraulic gradients, groundwater pH's, distances of containments from the nearest waterbodies, levels of watertables with respect to the containment floors, types of waterbodies closest to the containments, and slopes of surface waterbody floors. The relevant parameters were combined to estimate the average groundwater travel times from the reactor containments to the nearest waterbodies.

All sites with LWR's currently in operation and some other sites with reactors either under construction or proposed were included (Table B1 and Figure 4.2).[4] The information was obtained from the PSAR's, the FSAR's and certain environmental reports for the specific sites. No attempt was made to improve upon this information by consulting other sources. Where reasonable data was not given for some or all of the parameters, other available information was utilized to develop estimates of the missing parameters. Consequently, the assembled data should not be used in general for specific sites; rather, its only purpose is to aid in describing generic sites. (For specific information about a given site, one should use the appropriate SAR's, if adequate, and any more recent and/or complete data which might be available.)

Table B1. Selected Data on Nuclear Power Plants in the United States Operating, in Construction, or Proposed Through About 1985.[4]

Units and status: Number of letter equals number of units in given status: O = operating, C = construction, P = proposed.

Item	Name	Units & Status	State	Cooling Source	Power (net MWe)	Nearest Waterbody if Other Than Cooling Source
1	Allen's Creek	PP	Tex	Brazos River	2,300	dammed Lake
2	Arkansas	O,C	Ark	Arkansas River	1,762	Dardanelle Reservoir
3	Atlantic[a]	PPPP	N.J.	Atlantic Ocean	4,600	—
4	Bailly	C	Ind	Lake Michigan	660	—
5	Barton	PPPP	Ala	Coosa River	4,636	—
6	Beaver Valley	CC	Pa	Ohio River	1,704	—
7	Bellefonte	PP	Ala	Reservoir on Tennessee River	1,800	—
8	Big Rock Point	O	Mich	Lake Michigan	75	—
9	Black Fox	CC	Okla	Verdigris River	1,900	—
10	Blue Hills	PP	Tex	Sabine River	1,836	Mitchell Creek
11	Braidwood	PP	Ill	Kankakee River	2,240	—
12	Browns Ferry	OO,C	Ala	Lake Wheeler	3,195	—
13	Brunswick	CC	N.C.	Cape Fear Estuary	1,642	—
14	Byron	PP	Ill	Rock River	2,240	—
15	Callaway	PP	Mo	Missouri River	2,300	Auxvasse Creek
16	Calvert Cliffs	OO	Md	Chesapeake Bay	1,690	—
17	Catawba	PP	S.C.	Lake Wylie	2,306	—
18	Cherokee	PPP	S.C.	Broad River	3,840	—
19	Clinch River[b]	P	Tenn	Clinch River	350	—
20	Clinton	PP	Ill	Salt Creek	1,910	artificial Pond
21	Comanche Peak	PP	Tex	Squaw Creek Reservoir	2,300	—
22	Cook	O,C	Mich	Lake Michigan	2,120	—
23	Cooper	O	Nebr	Missouri River	778	—
24	Crystal River	O	Fla	Gulf of Mexico	825	—
25	Davis Besse	C,PP	Ohio	Lake Erie	2,718	—
26	Diablo Canyon	CC	Calif	Pacific Ocean	2,190	—
27	Douglas Point	PP	Md	Potomac River Estuary	2,356	—
28	Dresden	OOO	Ill	Illinois River	1,818	—
29	Duane Arnold	O	Iowa	Cedar River	569	—

Table B1 (cont)

Item	Name	Units & Status	State	Cooling Source	Power (net MWe)	Nearest Waterbody if Other Than Cooling Source
30	Farley, J.	CC	Ala	Woodruff Reservoir	1,658	—
31	Fermi	C,P	Mich	Lake Erie	2,303	--
32	Fitzpatrick	O	N.Y.	Lake Ontario	821	—
33	Forked River	C	N.J.	Barnegat Bay	1,070	—
34	Fort Calhoun	O	Nebr	Missouri River	457	—
35	Fort St. Vrain	O	Colo	South Platte River	330	St. Vrain Creek
36	Fulton	PP	Pa	Susquehanna River	2,280	Conowingo Pond
37	Ginna[c]	O	N.Y.	Lake Ontario	490	—
38	Grand Gulf	PP	Miss	Mississippi River	2,580	Lake Hamilton
39	Greenwood	PP	Mich	Lake Huron	2,400	Black River
40	Haddam Neck[d]	O	Conn	Connecticut River	575	—
41	Hanford	O	Wash	Columbia River	850	—
42	Hartsville	PPPP	Tenn	Cumberland River	4,820	Old Lake Hickory
43	Hatch Edwin	O,C	Ga	Altamaha River	1,581	—
44	Hope Creek[e]	PP	N.J.	Delaware Estuary	2,134	—
45	Humboldt Bay	O	Calif	Humboldt Bay	63	—
46	Indian Point	OO,C	N.Y.	Hudson Estuary	2,103	—
47	Jamesport	PP	N.Y.	Long Island Sound	2,300	—
48	Kewaunee	O	Wis	Lake Michigan	541	—
49	Koshkonong[f]	PP	Wis	Lake Koshkonong	1,800	—
50	LaCrosse	O	Wis	Mississippi River	50	—
51	LaSalle	CC	Ill	Kankakee River	2,156	Illinois River
52	Limerick	CC	Pa	Schuykill River	2,130	—
53	Maine Yankee	O	Maine	Back River Estuary	790	—
54	McGuire	CC	N.C.	Lake Norman	2,360	unnamed Creek
55	Midland	CC	Mich	Saginaw River	1,310	Tittabawassee River
56	Millstone	O,C,P	Conn	Niantic Bay	2,639	—
57	Montague	PP	Mass	Connecticut River	2,300	Green Pond
58	Monticello	O	Minn	Mississippi River	545	—
59	Nine Mile Pt	O,C	N.Y.	Lake Ontario	1,705	—
60	Norco	P	P.R.	Atlantic Ocean	583	—
61	North Anna	CCCC	Va	North Anna Reservoir	3,610	—
62	Oconee	OOO	S.C.	Lake Keowee	2,658	—
63	Oyster Creek	O	N.J.	Barnegat Bay	640	—
64	Palisades	O	Mich	Lake Michigan	700	—
65	Palo Verde	PPP	Ariz	Phoenix Sewage	3,810	Gila River
66	Peach Bottom	OOO	Pa	Lake Conowingo	2,170	—
67	Pebble Springs	PP	Oreg	Columbia River	2,520	Willow Creek
68	Perkins	PPP	N.C.	Yadkin River	3,840	—
69	Perry	PP	Ohio	Lake Erie	2,410	—
70	Pilgrim	O,C	Mass	Cape Cod Bay	1,844	—
71	Point Beach	OO	Wis	Lake Michigan	994	—
72	Prairie Island	OO	Minn	Mississippi River	1,060	Sturgeon Lake
73	Quad Cities	OO	Ill	Mississippi River	1,600	—
74	Rancho Seco	O	Calif	American River	804	Dry Creek
75	River Bend	PP	La	Mississippi River	1,868	—
76	Robinson	O	N.C.	Lake Robinson	700	—

Table B1 (cont)

Item	Name	Units & Status	State	Cooling Source	Power (net MWe)	Nearest Waterbody if Other Than Cooling Source
77	Salem(g)	CC	N.J.	Delaware Estuary	2,205	—
78	San Joaquin(h)	PPPP	Calif	Calif. Aqueduct	5,200	—
79	San Onofre	O,CC	Calif	Pacific Ocean	2,710	—
80	Seabrook	PP	N.H.	Atlantic Ocean	2,400	—
81	Sequoyah	CC	Tenn	Lake Chicamauga	2,240	—
82	Shearon Harris	PPPP	N.C.	Cape Fear River	3,660	reservoir
83	Shoreham	C	N.Y.	Long Island Sound	819	—
84	Skagit	PP	Wash	Skagit River	2,554	—
85	Somerset	PP	N.Y.	Lake Ontario	2,440	—
86	Sough Texas	PP	Tex	Colorado River	2,500	—
87	St. Rosalie(h)	PP	La	Mississippi River	2,320	—
88	Summer	C	S.C.	Lake Monticello	900	Broad River
89	Summit	PP	Del	Chesapeake-Delaware Canal	1,540	—
90	Surry	OO,PP	Va	James River Estuary	3,294	—
91	Susquehanna	CC	Pa	Susquehanna River	2,100	—
92	Sterling	P	N.Y.	Lake Ontario	1,150	—
93	St. Lucie(i)	CC	Fla	Atlantic Ocean	1,610	—
94	Three Mile Island	O,C	Pa	Susquehanna River	1,724	—
95	Trojan	C	Oreg	Columbia River	1,130	—
96	Turkey Point	OO	Fla	Biscayne Bay	1,386	—
97	Tyrone	PP	Wis	Chippewa River	2,300	—
98	Vermont Yankee	O	Vt	Connecticut River	514	—
99	Vidal	PP	Calif	Colorado River	1,540	—
100	Vogtle A.W.	CCCC	Ga	Savannah River	4,452	—
101	Waterford	P	La	Mississippi River	1,113	—
102	Watts Bar	CC	Tenn	Chickamauga Lake	2,338	—
103	Wolf Creek	P	Kans	Neosho River	1,150	cooling pond
104	WPPSS 1 & 4	PP	Wash	Columbia River	2,412	—
105	WPPSS 3 & 5	PP	Wash	Chehalis River	2,484	—
106	Yankee-Rowe	O	Mass	Lake Sherman	175	Deerfield River
107	Zimmer	C,P	Ohio	Ohio River	1,980	—
108	Zion	OO	Ill	Lake Michigan	2,100	—

[a] Floating plant - therefore not considered.
[b] Breeder reactor - therefore not considered.
[c] Alternative name is Brookwood.
[d] Alternative name is Connecticut Yankee.
[e] Alternative name is Newbold Island.
[f] Alternative name is Haven.
[g] Alternative name is Burlington.
[h] Not adequately documented.
[i] Alternative name is Hutchinson Island.

The under-construction and proposed sites were included in the compilation in an effort to improve the reliability of the characterization. For some of the earlier reactors, the information in the SAR's as to exactly what is under the containments is sketchy. However, the information available in the documents consulted has dramatically improved with the date of docketing. Therefore, some of the more recently documented LWR's were added to the list of reactors considered in the Reactor Safety Study.

To determine the appropriate range of hydrogeological parameters, the substratum underneath each containment was investigated. For naming purposes, the type of subsoil was approximated by the soil found immediately beneath the containment floor(s).* The fact that the melt debris might reside simultaneously in several thin layers of different composition was ignored in the determination of the soil type. For groundwater travel time estimations, the soil found beneath the containment floor was assumed to extend uninterrupted to the nearest waterbody. Inasmuch as the stratigraphy along the path to the nearest waterbody was not generally given, this seemed to be the only appropriate approximation. Such information would be required to perform an adequate study for a specific site. The types of substrata beneath the considered containments are listed in Table B2.

The permeability (or hydraulic conductivity) was taken to be as given. When not given, the permeability was approximated by either of three procedures: the permeability was taken to be that given for a very close site with a comparable stratigraphy but with more recent documentation; the permeability was taken to be approximately that for any site with an apparently comparable stratigraphy; or, the permeability was taken to be approximately equal to the field permeability for the specified rock type under the containment. All three methods resulted in comparable values if values at the high ends of the possible ranges were consistently utilized. For limestone, fracturing and solutioning were assumed if the documents consulted made no indication to the contrary. When the permeability was approximated for groundwater travel time estimations, it was taken to be that for the most permeable layer in which the melt debris might reside. The resulting distribution of the permeabilities for the considered sites is given in Figure B1.

Table B2. Rock Types Beneath LWR Sites

Rock Type	0 Ft Below	0 to 50 Ft Below
SEDIMENTARY		
Sand or sandstone	31	21
Conglomerate	1	0
Clay or claystone	5	4
Silt or siltstone	2	0
Limestone or dolomite	16	12
Shale	3	2
Outwash or terrace deposits	4	1
Lacustrine deposits (not till)	6	3
Till	6	1
Multiple sedimentary layers	12	40
Total Sedimentary	86	-
METAMORPHIC		
Gneiss	7	7
Schist	2	2
Slate	1	1
Total Metamorphic	10	-
IGNEOUS		
Basalt	1	1
Diorite	5	5
Agglomerate	1	0
Total Igneous	7	-

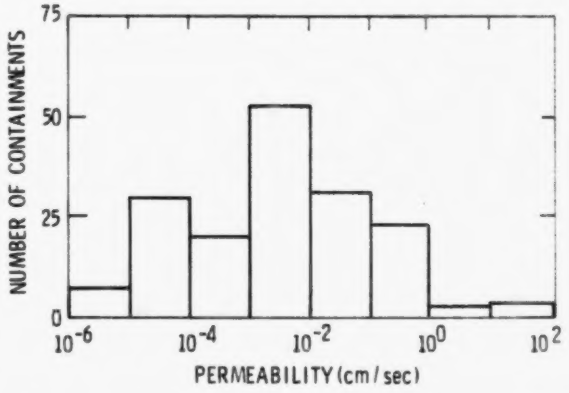

Figure B1. Distribution of Permeabilities for the Surveyed Containments

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*In contrast, for calculational purposes, the type of soil beneath each containment was approximated by the soil within 50 ft of the containment floor through which groundwater would move most rapidly. For details, see the discussion of the estimation of permeabilities later in this subsection.

The porosities were among the least often specified hydrogeological parameters and frequently had to be approximated. In particular, the effective porosities were given very rarely. When approximated for groundwater calculations, they were taken at the low ends of the possible ranges.

The groundwater gradient was taken along the steepest path to the nearest surface waterbody. The maximum groundwater level at the site was used to approximate the gradient if it was not given. If a groundwater level at the containment was not listed, it was approximated as the maximum possible from the given information; in those few cases, only the topography was available for such determinations and therefore was used to calculate the hydraulic gradient. The resulting distribution of hydraulic gradients for the considered sites is given in Figure B2.

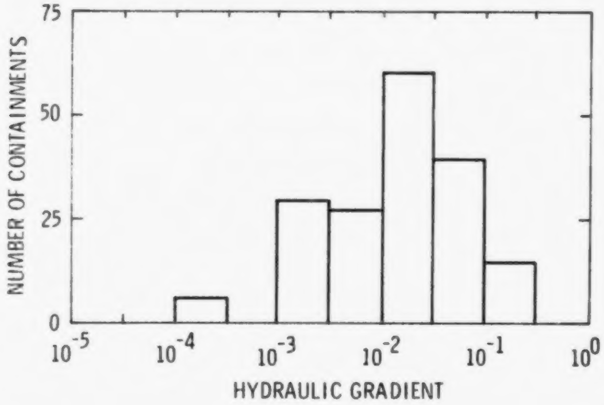
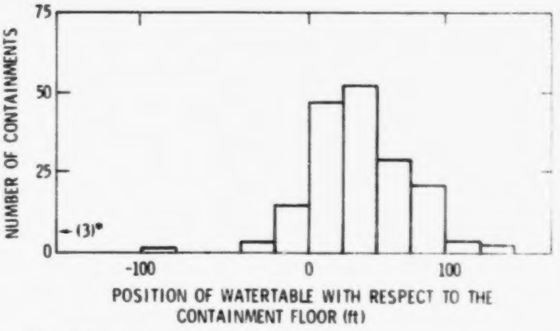


Figure B2. Distribution of Hydraulic Gradients for the Surveyed Containments

The position of the watertable relative to that of the containment floor was obtained using the elevation of the watertable as given. Possible changes in the position of the watertable due either to construction of the reactor buildings or to the development of related waterbodies were not taken into account unless such changes were considered in the consulted portions of the SAR's. The resulting spectrum of relative watertable positions for the considered sites is given in Figure B3.

For none of the reactor sites considered was any quantitative measure of dispersion given. However, dispersion could be very important (see Subsection 4.3). It is extremely site dependent. By ignoring the exact dispersion for a site, a path could be neglected by which the radionuclides could reach the waterbody much earlier than expected; alternatively, the difficulty that the water could encounter in reaching the waterbody might be significantly underestimated. Inasmuch as little information was supplied regarding the dispersive characteristics of the subsoil, a homogeneous isotropic medium was assumed.



*3 ENTRIES OFF THE FIGURE

Figure B3. Distribution of the Positions of the Watertables Relative to Those of the Containment Floors for the Surveyed Containments

The distance to the nearest waterbody was taken to be the shortest distance as measured down the hydraulic gradient. The nearest waterbody could not always be readily determined due to the poor quality of the maps in a few of the SAR's. When such difficulty was encountered, the distance was taken to the nearest evident waterbody. Furthermore, the changes in the hydraulic gradient due to construction of the reactor and especially due to the development of associated waterbodies were not always obvious. Consequently, the downgradient direction and therefore the closest waterbody were not always exactly determinable. Leakage to the surface via underground springs prior to subterranean contact with surface water was not considered. The resulting distribution of estimated distances to the nearest waterbodies from the considered containments is given in Figure B4.

B2.2.1.2 Usage

In the approximations employed in this study, the permeability, the hydraulic gradient, and the effective porosity can be used to estimate the average groundwater velocity. This velocity can, in turn, be combined with the downgradient distance from the site of the accident to the nearest waterbody to obtain the average groundwater travel time. This travel time can be utilized with the value of the appropriate distribution coefficient of any radionuclide to determine the average travel time of that nuclide to the nearest surface waterbody.

To characterize each reactor site, some of the information just reviewed was combined to estimate the average groundwater travel time for the site. The distributions of the distances to the nearest surface waterbodies, the magnitudes of the average groundwater velocities and the resultant average groundwater travel times for the considered reactors are given in Figure B4.

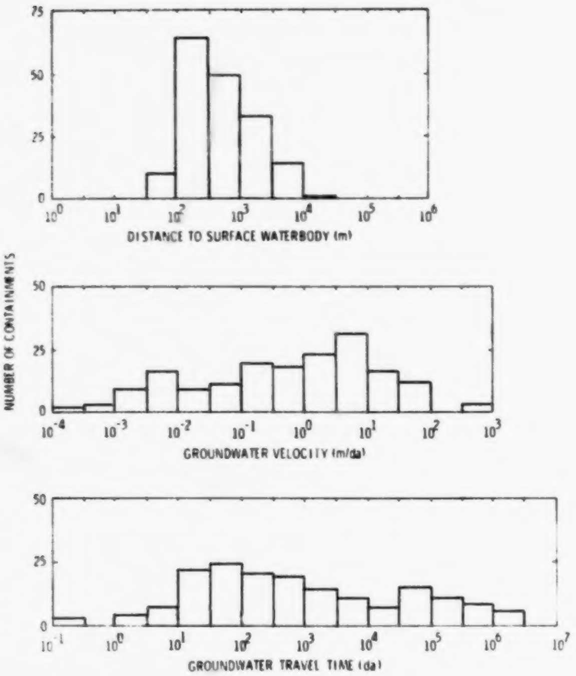


Figure B4. Distributions of the Distances to the Nearest Surface Waterbodies, the Groundwater Velocities and the Times Required for Groundwater to Reach the Nearest Surface Waterbodies for the Surveyed Containments

These travel times were, in turn, classified according to the nearest surface waterbody. Histograms of the travel times are presented in Figure B5 for large lake, estuary, ocean, "other lake," and river sites. The term "other lake" is used to denote an artificial reservoir on a river.

Those sites which had groundwater travel times of more than 10^5 days (approximately 274 years) were subsequently redefined to be dry sites. Only exposure via aquifer usage was considered at such sites. In practice, most of the sites defined to be dry would not experience any such usage (See Appendix E).

The goal of the compilation of hydrogeological data was to obtain a conservatively realistic set of groundwater travel times. The data for each site were always combined in a manner such that the lower limit of the range of indicated possible groundwater travel times for the site was obtained. However, the groundwater travel times presented in Figures B4 and B5 are not necessarily conservative.

For example, even though the melt debris resided in a substratum which was impervious, water from other more permeable strata could possibly reach that debris. Before the construction of any containment, much of the material originally around and beneath the area of the containment floor is disturbed. Thus, during the construction, channels can be opened in initially impervious layers through which water from higher (or lower) ground levels could possibly reach the melt debris. This might be especially important early in the release scenario when the heat of the melt could provide substantial convective transfer of contaminated water to higher, possibly more permeable, strata. Additionally, such channels could permit much more rapid release of sumpwater and containment gases into the environment than would normally be expected.

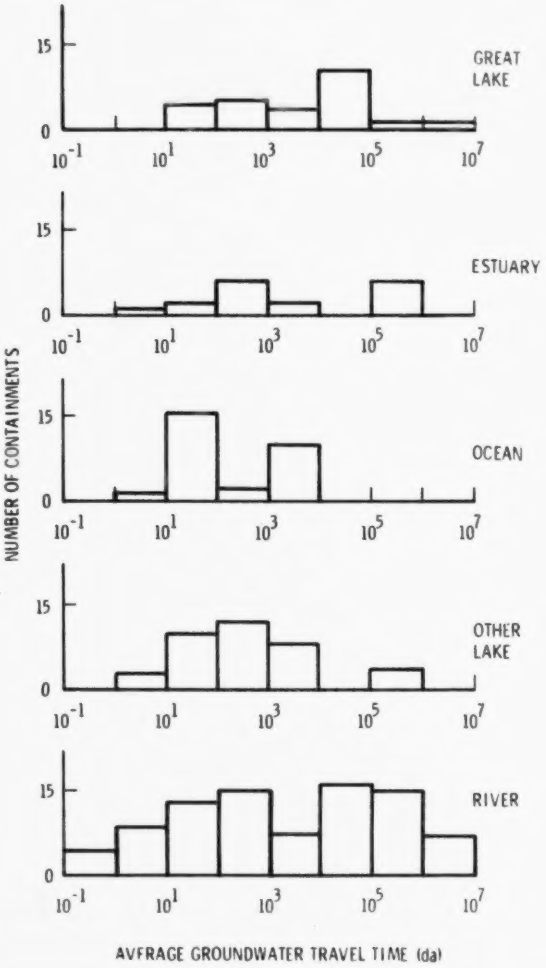


Figure B5. Groundwater Travel Times as a Function of the Nearest Surface Waterbody

The groundwater beneath a reactor site has practically no knowledge of what the nearest body of surface water is. Consequently, in a first approximation, groundwater transport can be modeled identically for the five distinct generic sites which have been chosen on the basis of surface-water hydrology. The major observable geohydrological distinctions between the different types of sites are expected to be the exact distributions of parameters appropriate to each type. Calculations were performed for an appropriate range of groundwater travel times for each type of site.

B2.2.2 Hydrological Parameters

Great variations exist in the possible parameters which might be used to describe aquifers and other subsurface waterbodies. Such waterbodies can range from a few meters in thickness to hundreds of meters or more. Effective porosities can range from much less than .001 to .30 or more. The hydraulic gradient and the permeability can result in groundwater velocities which range from less than a meter a year to more than a hundred meters per year.

For calculational purposes, the standard aquifer was defined to be 50 m in depth; the lateral and longitudinal extents were taken to be infinite. Except as mentioned previously, the effective porosity was taken to be .25. The longitudinal dispersion coefficient was taken to be 500 m^2/yr and the lateral dispersion coefficient was assumed to be 50 m^2/yr. None of the parameters just listed are considered to be necessarily representative; in general, they tend to be conservative for the purposes of this study.

As was explained in Subsection B2.2.1, the other parameters of importance and the "actual" effective porosity were lumped together for each site to form a single site characteristic, the average groundwater travel time. The travel times calculated from the results of the survey and used in this study are presented in Subsection B2.2.1. The distribution coefficients and the factors affecting them are discussed in the next subsection, B2.2.3.

B2.2.3 Distribution Coefficients

Chemical species in solution in groundwater can undergo many different reactions, both with other species in solution and with the soil through which they are passing. Among the important interactions are physical adsorption, ion exchange, precipitation, colloid formation, complex formation and fixation. The parameter which describes the resultant retardation of the movement of any nuclide relative to the movement of the groundwater is the previously-mentioned distribution coefficient, K_d. The addendum to this appendix describes the environmental factors affecting the values of this parameter.

The distribution coefficients can be as site dependent as the hydrological parameters. They depend upon the pH, the Eh (oxidation potential), the ionic strength and the chemical composition of both the groundwater and the soil. They also depend upon the exchange properties of the soil. The survey mentioned previously also included an attempt to obtain values for as many of these parameters as possible. In general, only the groundwater pH and composition were available at all, and these were noted for less than half the sites considered. All of the other chemical parameters which are needed to adequately describe the groundwater and the soil beneath the reactors were not contained in the surveyed documents.

The K_d's used in this study (Table B3) were obtained partially through laboratory experiments and partially through application of chemical scientific judgment based on the chemical processes involved.[5] The experiments in which these were obtained utilized "typical western desert soil" through which flowed "typical western desert groundwater."

This groundwater can be regarded as being typical of the groundwater found underneath most LWR's in at least two respects. First, its pH indicates that the water is alkaline; for those reactors for which information was available, the groundwater immediately beneath the containment is usually alkaline. Furthermore, the ionic strength of the desert groundwater is in the same range as the ionic strengths of the groundwaters under most of those reactor sites for which such information was recorded.

As was just mentioned, the groundwater at most reactor sites is alkaline. For those sites which have acidic groundwater, there could be some significant differences in the relative K_d's which would be appropriate. Among the notable variations might be relatively more rapid transport of the lanthanides and certain actinides. For elements such as these, the changes in the appropriate K_d's due to pH variations could be orders of magnitude. Although large variations in certain K_d's might be observed for any slight change in pH, the largest number of effects would probably be observed close to the transition from alkaline to acidic pH's, that is, for variations near pH = 7. Fortunately, those radionuclides for which the largest variations of K_d's might be expected tend to be minor contributors to the consequences and risks of releases to the hydrosphere after a meltdown accident.*

The values of the distribution coefficients employed for groundwater transport in this study are thought, in general, to be somewhat low. The use of such low distribution coefficients is conservative. At non-dry sites, relatively low values of the K_d's result in early emergence of the nuclides from the aquifer. Consequently, less decay is predicted to occur before exposure than would actually be the case. The larger amount of undecayed material which is predicted to reach the surface waterbody results in somewhat of an overestimation of the consequences. At a dry site, the lower K_d's result in a relatively higher percentage of the radioactive material being predicted to be in solution at any given time than would otherwise be the situation. As a result, low values for the distribution coefficients also result in an overestimation of the consequences.

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*See Chapter 6.

Because of the complex series of interactions and properties which determine the appropriate set of distribution coefficients for any specific site, it can be readily imagined that variations between sites can result in orders of magnitude differences in the coefficients appropriate for certain elements. Consequently, our estimates of the distribution coefficients can be out of line by orders of magnitude for certain sites. However, the values employed in this study probably tend to be conservative for most sites.

Table B3. Distribution Coefficients for Various Radionuclides in Groundwater

Radionuclide	$K_d^{(a)}$ (ml/g)	$K_d^{(b)}$ (ml/g)
H	0	0
Co		75
Ni		80
Rb	15	125
Sr	2	20
Y	200	2000
Zr	200	2000
Nb		2000
Mo	5	5
Tc	0.1	0
Ru	0 and 4	0 and 4[c]
Rh	25	250[c]
Ag		250[c]
Cd	25	2000
Sn	15	250
Sb	15	15
Te	20	20[c]
I	0.1	0
Cs	20	200
Ba	3	30[c]
La		600[c]
Ce	50	600[c]
Pr	60	600
Nd		600[c]
Pm	60	600[c]
Sm	60	600
Eu	60	600
U	100 (60)	3000
Np	60	15
Pu	200	2000
Am	100	2000
Cm	150	600

[a] Used in WASH-1400[6] and NUREG-0140.[7]

[b] Appropriate for "typical western desert soil" with "typical-western desert soil" groundwater flowing throughout it.

[c] Denotes estimated K_d.

Throughout this study, the distribution coefficients for the soil are among the least well known but most important parameters. The effects of varying these coefficients within reasonable bounds is considered in the section on sensitivity.

B3. Surface Water Transport

All the models used to describe surface water transport and dispersion and sediment-nuclide interactions are essentially those employed in the NRC Liquid Pathway Generic Study (NUREG-0440).[1] They are presented here for completeness. The modifications performed for this study and the differences in the ways in which they are utilized in the two studies are indicated.

As mentioned previously, the general movement of a contaminant in any waterbody can be described by Equation (B1-1). For certain types of releases, this equation can be solved analytically under the following simplifying assumptions:

1. all the coefficients in Equation (B1-1) are constants with respect to variations in both space and time;
2. the waterbodies have straight shorelines and are of uniform depth, or else the waterbodies are uniformly mixed; and,
3. the waterbodies are isotropic with respect to dispersal of contamination.

In this study, these assumptions have been employed for modeling lakes, estuaries and oceans. Consequently, for all surface waterbodies except rivers, the advantages of analytical solutions have been utilized. The general treatment for each of these waterbodies is analogous to that already presented for aquifers. For any waterbody residing in a series of other waterbodies, appropriate combinations of the spatially-varying analytical solutions and mixed-tank descriptions have been used. These solutions have been combined with integration techniques, as necessary, to account for complex release or efflux functions.

A river could be modeled in the same manner as the other types of waterbodies have been. However, adequate modeling of extended river systems generally requires the utilization of a rather long series of solutions. The advantage of analytic solutions can be lost in dealing with such a complex system. To adequately describe all the variations along a typical river, a different, more complex model has been developed. In that model, the river is divided into a series of compartments, each of which satisfies the above criteria utilized for the other waterbodies; variation of certain important coefficients is permitted between compartments. The resulting series of equations have been solved by employing methods typically utilized for dealing with differential equations.

B3.1 Large Lakes

B3.1.1 Models

All of the naturally-occurring lakes on which reactors reside are Great Lakes. Therefore, this discussion for lakes is restricted to the consideration of just the Great Lakes.

Three models are used in this study to consider movement of radionuclides within any Great Lake: nearshore, surface mixed-tank, and complete mixed-tank. The first describes local movement along the coast; the second describes mixing in the upper 50 to 75 feet of the lake; and the third describes the mixing in the complete lake due to seasonal turnover. A fourth model is used to consider the movement of radionuclides into successive waterbodies: a connected series of completely-mixed tanks. This model primarily describes movement into and within the other Great Lakes. Sediment effects are considered in all the models except the nearshore region one.

The concentration of a radionuclide in the coastal currents in the nearshore region (Figure B6) can be described by

$$\frac{dc}{dt} = - u_x \frac{dc}{dx} + E_x \frac{d^2c}{dx^2} + E_y \frac{d^2c}{dy^2} - \lambda c + S(t) \ , \qquad \text{(B3-1)}$$

where E_x and E_y are the alongshore and the offshore dispersion coefficients, respectively. An analytical solution is permitted if one assumes that the nearshore region has a constant depth d, a straight shoreline, constant dispersion coefficients and a constant alongshore water velocity of magnitude u_x.

Under these conditions, the concentration close to the shore resulting from an instantaneous release at the time $t = 0$ from a point source can be written as

$$c_i = Q_i \, c_{ix3} \, c_{iy3} \, c_{iz3} \, \Lambda_i \ , \qquad \text{(B3-2)}$$

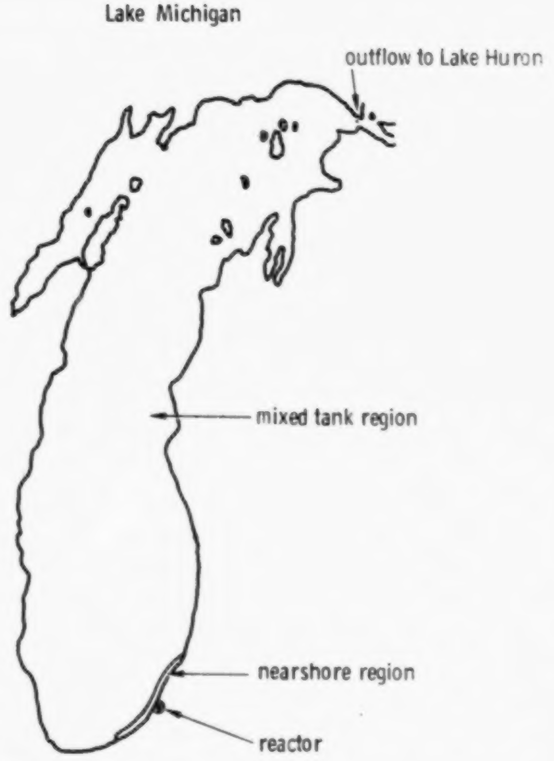

Figure B6. Nearshore and Mixed-Tank Regions for a Typical Great Lake

where

$$c_{ix3} = (4\pi E_x t)^{-1/2} \exp[-(x-ut)^2/4E_x t] \ ,$$

$$c_{iy3} = 2(4\pi E_y t)^{-1/2} \exp[-y^2/4E_y t] \ ,$$

$$c_{iz3} = (d)^{-1} \ , \qquad \text{(B3-3)}$$

and Λ_i is as defined previously. For a continuous release, the concentration close to the shore can be obtained by using the above solution for an instantaneous release in the convolution expression presented in Equation (B2-9). For a lake, as many for any other surface waterbody, the source term, S(t), represents essentially the appropriate amount of material passing across the aquifer-waterbody interface as a function of time.

The mixed-tank models for a lake can be described by

$$\frac{dc}{dt} = \frac{Q_i}{V} - \lambda_i c - \frac{q}{V} c \ , \qquad \text{(B3-4)}$$

where V is the volume of either the entire lake or just the upper layer of the lake and q is the flow rate out of the lake for water. For an instantaneous release at $t = 0$, the concentration in the lake is given by

$$c_i = (Q_i/V) \exp\{-[(q/V) + \lambda_i]t\} \ . \tag{B3-5}$$

In a zeroth-order approximation, the Great Lakes can be described as a series of completely-mixed reservoirs. For an instantaneous release into the first lake at $t = 0$, the concentration of the ith nuclide in the mth lake can be written as

$$c_{im} = \frac{Q_i \prod_{\ell=1}^{m-1} q_\ell}{\prod_{\ell=1}^{m} V_\ell} \cdot \frac{\sum_{j=1}^{m} \{-\lambda'_{ij} \exp(-\lambda'_{ij} t)\}}{\prod_{k=1,\neq j}^{m+1} (\lambda'_{ik} - \lambda'_{ij})} , \tag{B3-6}$$

where

$$\lambda'_{ij} \equiv (q_j/V_j) + \lambda_i \qquad \text{for } j \leq m \ ,$$

$$\lambda'_{ij} \equiv 0 \qquad \text{for } j > m \ . \tag{B3-7}$$

To consider continuous releases, either Equation (B3-5) or Equation (B3-6) can be used in the convolution integral in Equation (B2-9).

Simultaneously with transport by surface water movements, the radionuclides undergo interactions with the sediment particles (Figure B7). These interactions effectively remove some of the radionuclides from the water and therefore lower the concentrations of the nuclides in the water.

In the sediment-radionuclide interaction model,[1,8] the radionuclides residing on the suspended sediments are assumed to be in equilibrium with respect to the radionuclides in the water. Consequently, the concentration of a radionuclide residing on the suspended sediment, c_{ss}, can be determined from the appropriate distribution coefficient, K_d, and the concentration of the nuclide in the water, c_w, according to

$$c_{ss} = K_d \cdot c_w \ . \tag{B3-8}$$

As sedimentation occurs, the radionuclides residing on the sediment particles are removed from the water column at a rate

$$r_{sed} = c_{ss} (v\rho_s/d_w) = c_w K_d (v\rho_s/d_w) \ , \tag{B3-9}$$

where v is the sedimentation rate (depth/year), ρ_s is the density of the sediment in the sediment layer, and d_w is the depth of the water column (Figure B7).

In a similar fashion, the nuclides in the water between the nonsuspended sediment particles (in the interstitial water) are assumed to be in equilibrium with nonsuspended sediment. Consequently, Equation (B3-8) is assumed to hold separately for the suspended sediment and the bulk of the water, and for the deposited sediment and its associated interstitial water.

The concentration in the interstitial water can be different than the concentration in the water column. Therefore, a net diffusion of radionuclides can occur across the sediment-water interface. The net rate of flow is proportional to the net difference in the concentrations in the interstitial and the non-interstitial waters. The net rate of flow from the interstitial water is assumed to be[8-10]

$$r_{diff} = (K_f/d_w)[(c_{sd}/K_d) - c_w] \ , \tag{B3-10}$$

where c_{sd} is the concentration in the deposited sediment and K_f is the diffusional transfer rate (distance/time). K_f is approximately the apparent diffusion coefficient divided by the layer thickness.

The ordering of apparent diffusion coefficients according to their magnitudes is essentially the reverse of that for distribution coefficients. That is, the stronger the binding of a particular radionuclide to the sediment is, the smaller the apparent diffusion coefficient is and hence the slower the diffusion process is. However, in this report, the diffusional transfer rate is taken to be element independent.

If sedimentation occurs at a fast enough rate, then diffusion can be too slow to return the radionuclides from the sediment to the water column before decay. Therefore, the radionuclides can be effectively buried and lost to further contact with the water. To consider the effect of such removal by burial, a sediment depth below which the nuclides are effectively

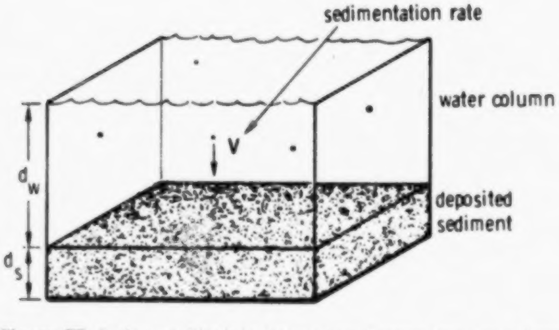

Figure B7. Sediment-Nuclide Interaction Parameters for a Large Lake

lost can be assumed. The rate of loss by such burial can be written as

$$r_{burial} = (v\rho_s/d_s)\, c_{sd} \,, \tag{B3-11}$$

where d_s is the thickness of interacting sediment. Inasmuch as the diffusion rate is element dependent, the depth at which any given element can be considered to be buried varies. However, in this report, the burial depth is taken to be element independent.

The resulting equations describing the distribution of the nuclides between the sediments and the noninterstitial water can be written as

$$\frac{dc_w}{dt} = \frac{Q}{V} - r_{flush} - r_{decay\ (water)} - r_{sed} + r_{diff} \,, \tag{B3-12}$$

$$\frac{dc_s}{dt} = - r_{decay\ (sed)} + r_{sed}\frac{d_w}{d_s} + r_{diff}\frac{d_w}{\rho_s d_s} + r_{burial} \,, \tag{B3-13}$$

where

$$r_{flush} \equiv (q/V)c_w \,,$$

$$r_{decay\ (water)} \equiv \lambda c_w \,,$$

$$r_{decay\ (sediment)} \equiv \lambda c_s \,,$$

and the factors (d_w/d_s) in the second equation result from the difference between the water and the sediment volumes.

Alternatively, these equations can be written as

$$\frac{dc_w}{dt} = \frac{Q}{V} + c_s\lambda_1 - c_w\lambda_2 \,,$$

$$\frac{dc_s}{dt} = c_w\lambda_3 - c_s\lambda_4 \,, \tag{B3-14}$$

where

$$\lambda_1 \equiv (K_f/d_wK_d) \,,$$

$$\lambda_2 \equiv (q/V) + \lambda + (v\rho_sK_d/d_w) + (K_f/d_w) \,,$$

$$\lambda_3 \equiv (vK_d/d_s) + (K_f/d_s\rho_s) \,,$$

$$\lambda_4 \equiv \lambda + (v/d_s) + (K_f/d_s\rho_sK_d) \,. \tag{B3-15}$$

For an instantaneous release at $t = 0$, the concentration in the water resulting from Equations (B3-8), (B3-12) and (B3-13) can be written as

$$c_i = [V(S_1-S_2)]^{-1}[(\lambda_4+S_1)\exp(S_1t) - (\lambda_4+S_2)\exp(S_2t)] \,, \tag{B3-16}$$

where

$$S_{1,2} \equiv \{-(\lambda_2+\lambda_4) \pm [(\lambda_2+\lambda_4)^2 - 4(\lambda_2\lambda_4 - \lambda_1\lambda_3)]^{1/2}\}/2 \,. \tag{B3-17}$$

For continuous releases, the convolution integral given in Equation (B2-9) must be utilized with Equation (B3-16).

For relatively high values of K_d, the concentration given by Equation (B3-16) is approximately

$$c_i \cong V^{-1}\exp(-\lambda_2 t) \,.$$

For relatively low values of K_d, the concentration is also given approximately by the same expression; however, unlike the high K_d situation, the K_d term in the expression for λ_2 is negligible. This expression for low values is essentially just the solution for the concentration presented in Equation (B3-5); in that equation, sediment interactions were completely neglected. The diffusion term is somewhat overestimated by this expression for small values of K_d.

The models just discussed can be combined so that the effects of sediment-nuclide interactions can be considered for series of reservoirs or lakes. For either relatively high or low values of K_d, the concentration of the ith nuclide in the mth lake can be written approximately as

$$c_{im} \cong \frac{Q_i \prod_{\ell=1}^{m-1} q_\ell}{\prod_{\ell=1}^{m} V_\ell} \cdot \frac{\sum_{j=1}^{m}\{-\lambda'_{ij}\exp[-\lambda'_{ij}t]\}}{\prod_{k=1,\,\neq j}^{m+1}[\lambda'_{ik} - \lambda'_{ij}]} \,,$$

where

$$\lambda'_{ij} \equiv (q_j/V_j) + (vK_d/d_w) + \lambda_i \qquad \text{for } j \le m \,,$$

$$\lambda'_{ij} \equiv 0 \qquad \text{for } j > m \,.$$

B3.1.2 Parameters

In this report, Lake Michigan is taken to be the generic large lake.

B3.1.2.1 Hydrological Parameters

As is shown in Subsection B4.4.1, the choice of Lake Michigan can be easily justified on the basis of hydrological considerations. Such justification is possible because the hydrological parameters for the Great Lakes are readily available.

The hydrological parameters for the Great Lakes are probably the best determined of the parameters for the various types of sites. This is due jointly to the limited number of Great Lakes, to the large number of studies which have occurred on these important waterbodies, and to the small variation in parameters which occurs among the lakes.

The volumes, outflow rates, average residence times, surface areas, maximum depths, and average depths for the Great Lakes are as presented in Table B4.[11,12] For all mixed-tank calculations, the appropriate average depths were employed. For the nearshore calculations, the average depth was taken to be 10 m. In all the lakes, representative values for the dispersion coefficients are 500 to 1000 cm^2/sec for E_x, and 1 to 30 cm^2/sec for E_y;[13] typical values for the alongshore current u_x are 10 to 20 cm/sec.[13] For most calculations, the dispersion coefficients were taken as their minimum values and u_x was taken to be 10 cm/sec (8.64 km/da). Representative values for the net sedimentation rate, v, are 3×10^{-4} to 8×10^{-4} m/yr; typical values for the diffusion rate, K_f, are 0.1 to 1.6 m/yr.[9] For most calculations, v was taken as 5×10^{-4} m/yr and K_f was taken as 0.4 m/yr.

B3.1.2.2 Distribution Coefficients

The appropriate values for the sediment distribution coefficients are among the least well identified parameters for a Great Lake site. These parameters depend upon the same physical and chemical properties as the distribution coefficients for soil. These properties are discussed in detail in the addendum of this appendix. In this study, these parameters were always taken to be toward the conservative end of their possible ranges; that is, they were taken to be low. The appropriate values of the distribution coefficients for Great Lake sediment were taken to be 50 times the values employed in the soil (Table B3). For example, for Sr and Cs, this results in values of 10^3 and 10^4, respectively.

B3.2 Estuaries

B3.2.1 Model

A one-dimensional sectionally-averaged model is employed here to consider all estuaries.[14,15] Tidal influences are taken to be adequately described by a "tidally-averaged" approximation in which the tidal oscillations are not included explicitly. Instead, they are taken to be sufficiently described by longitudinal dispersion. In such a model, the concentration of a radionuclide in an estuary with net downstream flow and accompanying tidally-induced dispersion can be given by

$$\frac{dc}{dt} = - u_x \frac{dc}{dx} + E_x \frac{d^2c}{dx^2} - \lambda c + S(t) \ , \qquad \text{(B3-18)}$$

where u_x is the net downstream water speed and E_x is the longitudinal dispersion coefficient. An analytical solution is permitted if one assumes that the estuary has a constant depth d, a straight shoreline, a constant dispersion coefficient, and a constant net downstream velocity. Under these conditions, the concentration resulting from an instantaneous release at t = 0 from a point source at the origin can be written as

$$c_i = A^{-1}(4\pi E_x t)^{-1/2} \exp\{-[(x-ut)^2/4E_x t + \lambda t]\} \ , \qquad \text{(B3-19)}$$

where A is the cross-sectional area. For a continuous release, the concentration can be obtained by using Equation (B3-19) in the convolutional integral presented in Equation (B2-9).

Simultaneously with the transport of radionuclides by water movements, the radionuclides undergo interaction with the sediments (Figure B5). Basically, the sediment-nuclide interaction model employed for estuaries (Figure B8) is the same as that utilized for large lakes, with the following major difference: the frequently resuspended sediments are assumed to form a top layer of sediment which is in equilibrium with the nuclides in the water column. In addition, diffusion back into the water column is ignored due to the relatively high sedimentation rate found in most estuaries.

The equation describing the resulting concentration in the water phase can be written as

$$\frac{dc}{dt} = - u'_x \frac{dc}{dx} + E'_x \frac{d^2c}{dx^2} - \lambda c - r_{burial} \ , \qquad \text{(B3-20)}$$

where

$$u'_x = [fu_x + (1-f)u_b\rho_s n_s K_d]/[f + (1-f)\rho_s n_s K_d] \ ,$$

$$E'_x = [fE_x + (1-f)E_b\rho_s n_s K_d]/[f + (1-f)\rho_s n_s K_d] \ ,$$

$$f = d_w/(d_w + d_s) \ . \qquad \text{(B3-21)}$$

In these expressions, u_b is the net downstream speed of the frequently resuspended sediment; f is the fraction of interaction space occupied by non-interstitial water and (l-f) is the fraction of space occupied by sediment available for interaction; ρ_s is the density of the sediment assuming no porosity; and n_s is its porosity in the sediment layer. Therefore, $\rho_s n_s$ is the density of the sediment in the sediment layer. In the limit of no resuspension, f approaches l; consequently, u'_x approaches u_x, and E'_x approaches E_x.

Table B4. Parameters for the Great Lakes

Lake	Volume (m^3)	Outflow (m^3/yr)	Residence Time (yr)	Surface Area (m^2)	Maximum Depth (m)	Average Depth (m)
Superior	1.22×10^{13}	6.50×10^{10}[(a)]	189	2.48×10^{10}	397	150
Michigan	4.87×10^{12}	1.57×10^{11}[(a),(b)]	31.0	1.74×10^{10}	281	75
Huron	3.80×10^{12}	1.57×10^{11}	24.2	1.99×10^{10}	229	60
Erie	4.58×10^{11}	1.73×10^{11}	2.65	7.72×10^{9}	64	15
Ontario	1.64×10^{12}	2.07×10^{11}	7.92	5.86×10^{9}	237	90

[a] Superior and Michigan both flow into Huron.
[b] Other estimates of this value are as low as 5×10^{10} m^3/yr.

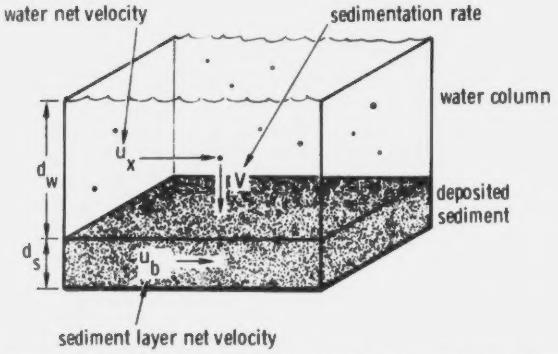

Figure B8. Sediment-Nuclide Interaction Parameters for an Estuary

For instantaneous release at t = 0, the concentration in the water resulting from Equation (B3-20) can be written as

$$c_i = (aA)^{-1} \ (4\pi E_x' t)^{-1/2} \ \exp\{-[(x-u_x' t)^2/4E_x' t + \lambda t]\}$$

$$\exp\{-vK_d t/d_w\} \ , \qquad \text{(B3-22)}$$

where

$$a \equiv f + (1 - f)\rho_s n_s K_d . \qquad \text{(B3-23)}$$

It is interesting to compare this basic expression for the concentration in an estuary to that for the concentration in an aquifer. To aid in this comparison, it should be noted that the expression $\rho_s n_s (d_s/d_w)$ can be written as $\rho_{bs}(\text{"n"})^{-1}$, where ρ_{bs} is the bulk density of the sediment and "n" is its effective porosity. For both of these parameters, the volume is taken as that of the interacting sediment plus that of all of the water. Thus, by utilizing this expression in the equation describing the concentration in an estuary, it can be seen that an estuary can be considered to be like an aquifer with an extremely high porosity, provided burial is negligible.

Therefore, the net effects of the sediment interactions in an estuary are to retard the downstream movement of some of the radioactivity by repeated sedimentation and resuspension and to remove some of the rest of the radioactivity by burial. The proportion of water to available sediment in an estuary is much larger than the proportion of water to soil in an aquifer; as a result, the net retardation of the movement of the radioactivity is much less in an estuary than in the ground, if the same distribution coefficients (K_d's) are used. However, the effects of adsorption and other delaying processes can still be significant in an estuary.

For continuous releases, Equation (B3-22) can be used in the convolution integral in Equation (B2-9). For a release of the form be^{-bt} from a point source into an estuary, this produces

$$c(t) = \int_0^t be^{-bt'} \ c_{instant}(t-t') \ dt'$$

$$= (aA)^{-1} \ (4E_x')^{-1/2} \ b \ \exp[-\beta] \ (2\alpha^{1/2})^{-1} \ .$$

$$\{\exp[2(\alpha\gamma)^{1/2}] \ \mathrm{erfc}[(\gamma/t)^{1/2}+(\alpha t)^{1/2}]$$

$$+ \ \exp[-2(\alpha\gamma)^{1/2}] \ \mathrm{erfc}[-(\gamma/t)^{1/2}+(\alpha t)^{1/2}]$$

$$+ \ 2\exp[-2(\alpha\gamma)^{1/2}]\} \ , \qquad \text{(B3-24)}$$

where

$$\alpha \equiv [u_x'^2/(4E_x') + \lambda - b] \ ,$$

$$\beta \equiv [-xu'/(2E_x') + bt] \ ,$$

$$\gamma \equiv [x^2/4E_x'] \ . \qquad \text{(B3-25)}$$

erfc denotes the unit complement of the error function[1] and is defined in Equation (B2-12).

B3.2.2 Parameters

In this report, two "generic" estuaries are considered. They are based loosely on the Chesapeake Estuary and the Delaware Estuary.

B3.2.2.1 Hydrological Parameters

The hydrological parameters appropriate for a representative estuary are not nearly as easily determined as those for a typical large lake. Determination of the appropriate range for each of the parameters is beyond the scope of this study. Furthermore, account would need to be taken of the differences between the types of estuaries, whereas this report considers only one type of estuary. In view of these considerations, we do not suggest that the two estuaries which are utilized in this report are the ideal generic estuaries. Rather, they are possible estuaries. The subsection which deals with sensitivities discusses how variations in the parameters change the results so that the reader can judge the net effects of any changes which he or she might want to impose.

The magnitudes of the net water velocities of both the estuaries were taken to be 2.00 x 10^5 m/yr; the longitudinal dispersion coefficients of both were taken to be 10^9 m/yr. The average depths of the water columns were taken to be 8.54 m and 5.00 m, respectively; the maximum depth of the interacting sediment was taken to be 0.01 m. Burial below 0.01 m was considered to remove the radioactivity from all future interactions. The sedimentation rates for both were taken to be 2 x 10^{-3} m/yr.[16-18] The cross-sectional areas of the estuaries were 3.3 x 10^5 m^2 and 5.0 x 10^4 m^2, respectively; lengths of 300 km and 200 km, respectively, were considered, with 50 km above the source of radioactivity and the rest below.

B3.2.2.2 Distribution Coefficients

The sediment distribution coefficients appropriate for a representative estuary are again not easily estimated parameters. As in the case of the Great Lakes, this is due, in part, to the unspecified chemistry of the estuary water and sediments. It is also caused by the varying salinity of the water which is found as one proceeds down an estuary. Inasmuch as the distribution coefficients of some elements are strong functions of salinity, these coefficients can vary by orders of magnitude with distance along the estuary. However, this sophistication is beyond the data currently available for most elements. Distribution coefficients thought to be appropriate for the dominant radionuclides in marine conditions are utilized throughout the estuary. In the model employed in this study, the distribution coefficients are taken to be 5 times the values employed in the soil. For example, for Sr and Cs, this results in values of 10^2 and 10^3, respectively.

B3.3 Oceans

B3.3.1 Models

In an ocean, the initial water movement in open coastal water resembles that in the nearshore waters in the Great Lakes. As a result, the concentration of a radionuclide in the nearshore coastal currents can be described by

$$\frac{dc}{dt} = - u_x \frac{dc}{dx} + E_x \frac{d^2c}{dx^2} + E_y \frac{d^2c}{dy^2} - \lambda c + S(t) , \tag{B3-26}$$

where E_x and E_y are the alongshore and the offshore dispersion coefficients, respectively. Analytical solution is permitted if one assumes that the nearshore region has a constant depth d, a straight shoreline, constant dispersion coefficients, and a constant alongshore velocity of magnitude u_x (Figure B9).

Under these conditions, the concentration is given by the same expression as for the nearshore region in a Great Lake. That is, the solution for an instantaneous release from a point source at the origin at t = 0 is as given in Equations (B3-2) and (B3-3). For continuous releases, the convolution integral presented in Equation (B2-9) can be employed.

After any radioactivity leaves the nearshore region, it is assumed to move to the offshore region of the continental shelf. This region is taken to be described by a mixed tank (Figure B9). Flow out of the tank is taken to be into the deep portions of the ocean.

Inasmuch as the flushing times on the continental shelves are relatively short and the sedimentation rates are generally low, the effects of sedimentation can be ignored in a first-order approximation.

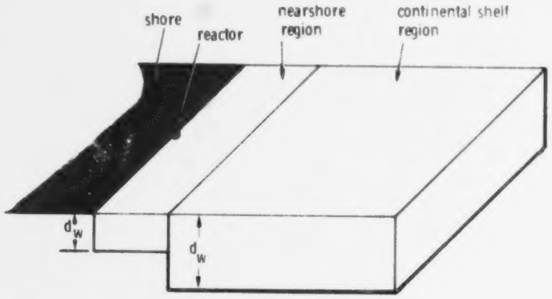

Figure B9. Nearshore and Mixed-Tank Regions for an Open Coastal Site

B3.3.2 Parameters

In this report, an arbitrary nonestuarial region of continental shelf is taken to be the generic ocean site.

B3.3.2.1 Hydrological Parameters

The hydrological parameters appropriate for a representative ocean site are not known. The required determination of the appropriate range for each of the parameters is beyond the scope of this study. In a more detailed study, account should be taken of the great differences that exist between ocean sites. Primarily this report considers an open coastal site which is perhaps hydrologically representative of certain Atlantic sites.[19,20] The effect of initial release into a relatively sheltered region is not dealt with. The impact of such a release might be significant because it could result in substantial population exposure in addition to that determined here.

The magnitude of the velocity of the nearshore current was taken to be 3.15 x 10^6 m/yr; the alongshore and the offshore dispersion coefficients were taken to be 3.15 x 10^6 m^2/yr and 3.15 x 10^5 m^2/yr, respectively. The average depth of the nearshore region was taken to be 10 m; the width and the length were taken to be 5 km and 400 km, respectively.

The offshore compartment was taken to be 50 m by 100 km by 400 km. The average residence time of the water was taken to be 2 years. Movement from the offshore compartment was assumed to be to the deep portions of the ocean.

B3.3.2.2 Distribution Coefficients

Sediment interactions are not considered in these models.

B3.4 River Systems

B3.4.1 Model

Transport of radionuclides by rivers has been extensively studied.[21-27] In general, the models for describing that transport require numerical solution of the differential equations. Codell,[1] by calling on a number of assumptions, was able to use simpler analytical solutions in the NRC Liquid Pathway Generic Study. A new numerical model, combining ease of use with adequate attention to detail, was developed specifically for this study. This model generally follows that of Codell, but includes some additional complexity.

The concentration of a radionuclide in flowing water in an element fixed in space in which the concentration is uniformly distributed in depth and lateral extent can be described by

$$\frac{dc_w}{dt} = \frac{d}{dx}\left(E_x \frac{dc_w}{dx}\right) - u\,\frac{dc_w}{dx} + \sum_{j=1}^{n} S_j(t) \quad \text{,(B3-27)}$$

where c_w is the concentration of activity per unit volume, E_x is the longitudinal dispersion coefficient, u is the magnitude of the local water velocity, and the set of S_j are the sources and sinks within the element. The concentration c_w includes the radioactive material dissolved in the water plus that sorbed to sediments traveling with the water. It is assumed that the longitudinal velocity of sediments carried by the water is equal to the water velocity.

The concentration in the bottom sediments (ignoring the possibility of sediment transport along the river system) can be taken to be described by

$$\frac{dc_s}{dt} = \sum_{j=1}^{n} S_j \ , \qquad \text{(B3-28)}$$

where c_s is the concentration in the bottom sediments. The sediment-nuclide interaction model employed for river systems is essentially identical to that utilized for large lakes. As a result, the sources and sinks are the same as those described in Subsection B3.1.

These equations were approximated by a combination of finite difference equations. Two typical finite difference elements are shown in Figure B10. The finite difference equations are

$$\begin{aligned} c^{j+1}_{w_i} = c^j_{w_i} + \Delta t \Bigg\{ & E_{xi}\left[\frac{c^j_{w_{i+1}} - c^j_{w_i}}{\Delta x_{i+1/2}} - \frac{c^j_{w_i} - c^j_{w_{i-1}}}{\Delta x_{i-1/2}}\right] \\ & + u^j_i \frac{c^j_{w_{i-1}} - c^j_{w_i}}{\Delta x_i} - \left[\lambda + \frac{v_{si}K_d + K_f}{d_{w_i}}\right] c^j_{w_i} \\ & + \frac{K_f}{K_d d_{w_i}} c^j_{s_i} \Bigg\} \end{aligned} \qquad \text{(B3-29)}$$

$$\begin{aligned} c^{j+1}_{s_i} = c^j_{s_i} + \Delta t \Bigg\{ & \left[\frac{v_{s_i}K_d + K_f}{d_{s_i}}\right] c^j_{w_i} \\ & - \left[\lambda + \left(v_{s_i} + \frac{K_f}{K_d}\right) d_{s_i}\right] c^j_{s_i} \Bigg\} , \end{aligned} \qquad \text{(B3-30)}$$

where $c^j_{w_i}$ is the concentration of the radionuclide in element i at time j;

$$\Delta x_{i+1/2} = \frac{(\Delta x_i + \Delta x_{i+1})}{2};$$

$$\Delta x_{i-1/2} = \frac{(\Delta x_i + \Delta_{i-1})}{2};$$

v_{s_i} is the net rate of sedimentation; d_{w_i} is the water depth; d_{s_i} is the effective sediment depth; $c^j_{s_i}$ is the concentration in the bottom sediment under element i at time j; and the other parameters are as previously defined. In this formulation, the transport term is in the Lelevier form,[28] whereas the dispersion term is in the classical form for the diffusion equation. The time step must satisfy the relationships $|u|\Delta t/\Delta x \leq 1$; $2E_x\,\Delta t/(\Delta x)^2 \leq 1$.

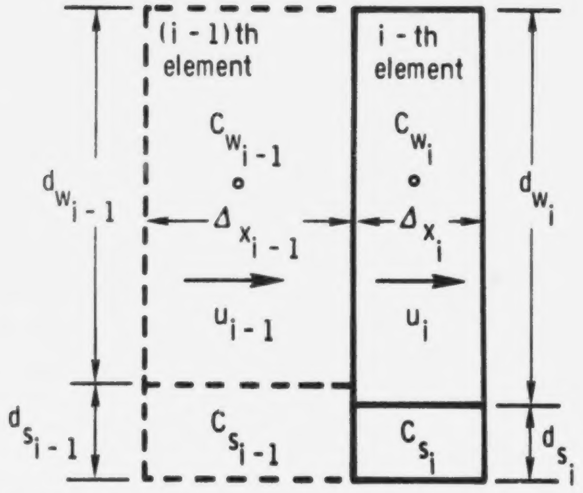

Figure B10. Two Typical Finite Difference Elements

In the model, flows can be treated as functions of time, thus allowing consideration of annual flood and low-water periods. The details of the procedure are presented in Section B3.4.2.1 and in Addendum B2.

B3.4.2 Parameters

In this report, two "generic" river systems are considered. They are based on the Tennessee-Ohio-Mississippi system and the Missouri-Mississippi system.

B3.4.2.1 Hydrological Parameters

The definition of a generic river system is not easy. Probably greater variation exists among the hydrological parameters which are important for describing river systems than exists among the parameters which are significant for considering any other type of waterbody. Unfortunately, more reactors are located on rivers than are located near any other type of surface waterbody.

Not only are the variations of parameters among different rivers substantial; the variations even within a given river can be significant. For example, flow rates can be highly variable. Seasonal variations in precipitation and run-off can result in changes of the appropriate flow rates and related flow volumes of over an order of magnitude. Furthermore, along the length of a given system, the rates and volumes can easily vary by at least an order of magnitude. Thus, a given river can exhibit a wide variation in appropriate hydrological parameters.

River systems often include both free-flowing regions and dammed sections. Most river systems which have LWR's sitting somewhere along their banks include one or more reservoirs. Consequently, the generic river system should probably have both dammed regions and free-flowing stretches.

The two river systems considered here, the Tennessee-Ohio-Mississippi (TOM) system and the Missouri-Mississippi (MM) system, have been chosen as being "representative" and are utilized for all the calculations. The former is typical of river systems containing reservoirs, whereas the latter represents river systems which do not have reservoirs. Both systems have LWR's located along their banks. They differ from other river systems having LWR's primarily in three respects: they are each longer than the typical river systems; they are each larger, at least in their lower reaches, than the typical systems; and they both flow into the Gulf of Mexico.

Reservoirs on the Tennessee system were modeled as rectangular tanks. Long reservoirs were modeled as a set of connected tanks of uniform width and length but varying depth. For multi-element reservoirs, depths were assumed to increase uniformly toward the dam end.

Major tributaries in both systems were appropriately located in the model. Minor tributaries were collected into a few discrete points.

The short stretch of the Tennessee between Kentucky Lake Dam and the confluence with the Ohio, and the short stretch of the Ohio included in the TOM system were each modeled as single sections. The Missouri and Mississippi were broken into segments of roughly 250 km each by dilution points representing tributaries. Each segment was additionally broken into 5-10 meshes.

The configuration used to consider the Tennessee-Ohio-Mississippi system is shown schematically in Figure B11. That utilized to describe the Missouri-Mississippi system is represented in Figure B12.

The details of the data analysis for the two river systems are given in Addendum B2 of this appendix. The salient features of this analysis are reviewed briefly here.

Data on flow, reservoir volume, etc., were taken from a variety of sources.[29-36] Where conflicts were found in the source data, estimates were made of data reliability and, as much as possible, the most reliable data were chosen.

For each reservior, the known back-water length* and surface area were used to compute an average width. The pool volume and surface area were used to compute an average depth.

---

*The backwater length is the distance from the dam to the point at which the original river slope intersects the reduced slope of the reservoir surface.

Sedimentation rates for the Tennessee River reservoirs were calculated from the data in Table B5, except that the rate for Hale's Bar was modified. Although the data indicated a net removal of sediment for that reservoir, a low positive rate of 2 mm/yr was assumed. A sedimentation rate of 0.1 mm/yr was arbitrarily used for all free-flowing stretches of the two river systems.

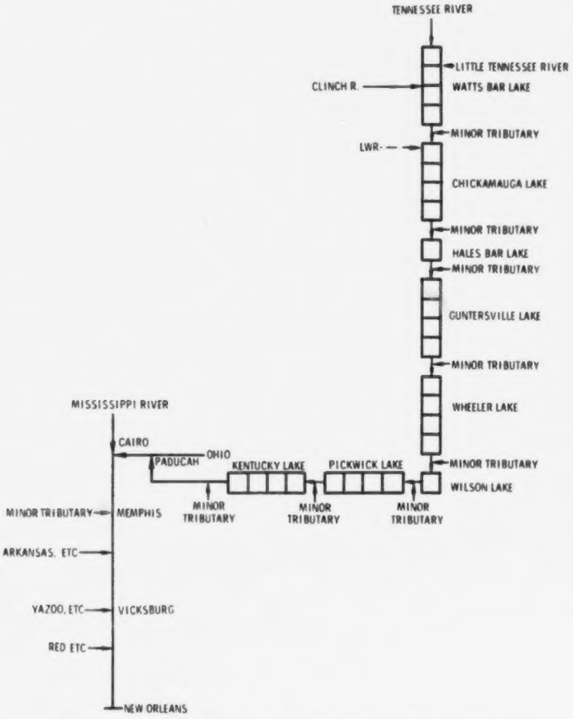
TENNESSEE RIVER
LITTLE TENNESSEE RIVER
CLINCH R.
WATTS BAR LAKE
MINOR TRIBUTARY
LWR
CHICKAMAUGA LAKE
MINOR TRIBUTARY
HALES BAR LAKE
MINOR TRIBUTARY
GUNTERSVILLE LAKE
MINOR TRIBUTARY
WHEELER LAKE
MINOR TRIBUTARY
WILSON LAKE
MISSISSIPPI RIVER
CAIRO
OHIO
PADUCAH
KENTUCKY LAKE
PICKWICK LAKE
MINOR TRIBUTARY
MINOR TRIBUTARY
MINOR TRIBUTARY
MINOR TRIBUTARY
MEMPHIS
ARKANSAS, ETC
YAZOO, ETC
VICKSBURG
RED ETC
NEW ORLEANS

Figure B11. Tennessee-Ohio-Mississippi Surface Water Model

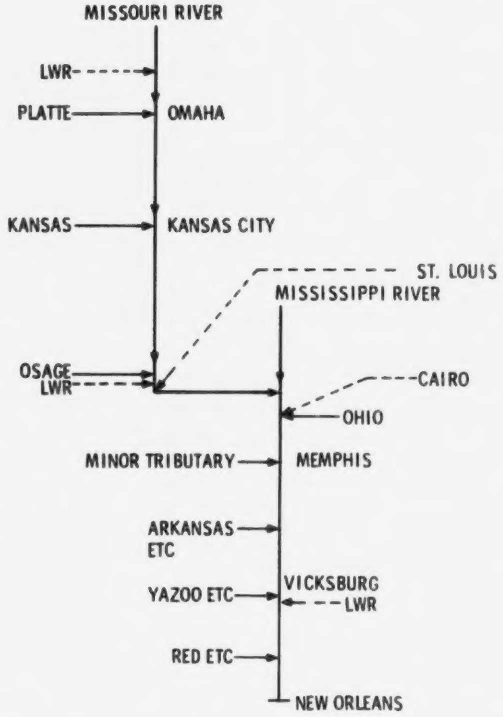
MISSOURI RIVER
LWR
PLATTE
OMAHA
KANSAS
KANSAS CITY
ST. LOUIS
MISSISSIPPI RIVER
OSAGE
LWR
CAIRO
OHIO
MINOR TRIBUTARY
MEMPHIS
ARKANSAS ETC
YAZOO ETC
VICKSBURG
LWR
RED ETC
NEW ORLEANS

Figure B12. Missouri-Mississippi Surface Water Model

The data relating to the sediment loads[31,32] of the free-flowing stretches of these river systems are discussed entirely in Addendum B2. The load-related information is not utilized in the river model but is included in this appendix for completeness. It is used in the pathway model involving the treatment of drinking water taken from rivers and is discussed further in Appendix D.

Table B5. Sedimentation in Tennessee River Reservoirs

Reservoir	Total Sediment Volume (acre-ft)	Time of Sedimentation (yr)	Net Sedimentation Rate (m/yr)
Watt's Bar	22,082	15.2	.0103
Chickamauga	7,781	21.1	.0029
Hale's Bar	-519 (scour)	25.7	(see text)
Guntersville	33,153	20.6	.0069
Wheeler	71,697	24.8	.0129
Wilson	46,372	32.6	.0271
Pickwick	25,057	23.6	.0069
Kentucky	88,694	15.1	.0068

In general, the width, the depth and the magnitude of the velocity of any free-flowing stretch of river were obtained by using empirically-derived equations relating each of these quantities to the flow rate.[30,37] Likewise, to investigate seasonal variations, each of these three was estimated from equations relating each to the ratio of the actual flow rate to the seasonally-averaged flow rate.[30,31,37]

The values employed for the dispersion coefficients were obtained using the velocity-dependent simplified Fischer formula.[38] The consideration of hydrological dispersion in rivers is discussed further in both Addenda B2 and B3.[33,38]

The parameters used for the Tennessee-Ohio-Mississippi system are summarized in Table B6. The parameters employed for the Missouri-Mississippi system are given in Table B7.

Table B6. Parameters of Tennessee-Ohio-Mississippi System

Length km	Volume m^3	Flow Rate m^3/yr	Velocity m/yr	Transit Time, yr	Net Sedimentation Rate, m/yr	Surface Area m^2	
31.82	.4789E9[a]	.2072E11	(b)	.0231	.0103		Watt's Bar Lake
29.00	.6110E9	.2072E11	(b)	.0295	.0103		
24.25	.7886E8	.2325E11	(b)	.0034	.0029	3.5345E7	Chickamauga Lake
23.25	.1683E9	.2325E11	(b)	.0072	.0029	3.9432E7	
23.25	.2576E9	.2325E11	(b)	.0111	.0029	3.9432E7	
23.25	.3470E9	.2325E11	(b)	.0149	.0029	3.9432E7	
64.20	.1823E9	.3314E11	(b)	.0055	.002	2.6607+7	Hale's Bar Lake
32.78	.1303E9	.3617E11	(b)	.0036	.0069	6.5134+7	Guntersville Lake
32.78	.2345E9	.3617E11	(b)	.0065	.0069	6.5134+7	
32.78	.3387E9	.3617E11	(b)	.0094	.0069	6.5134+7	
32.78	.4429E9	.3617E11	(b)	.0122	.0069	6.5134+7	
29.56	.1373E9	.3919E11	(b)	.0035	.0129	6.8638+7	Wheeler Lake
29.56	.2814E9	.3919E11	(b)	.0072	.0129	6.8638+7	
29.56	.4256E9	.3919E11	(b)	.0109	.0129	6.8638+7	
29.56	.5697E9	.3919E11	(b)	.0145	.0129	6.8638+7	
23.94	.7680E9	.4221E11	(b)	.0182	.0271	6.1933+7	Wilson Lake
20.95	.3578E8	.4524E11	(b)	.0008	.0069	1.7981+7	Pickwick Lake
20.95	.9679E8	.4524E11	(b)	.0021	.0069	1.7981+7	
20.95	.1576E9	.4524E11	(b)	.0035	.0069	1.7981+7	
20.95	.2186E9	.4524E11	(b)	.0048	.0069	1.7981+7	
20.14	(c)	.4762E11	.5953E8	.0003	.0001		Tennessee River
69.10	.5281E9	.4762E11	(b)	.0111	.0068	2.6403+8	Kentucky Lake
69.10	.1410E10	.4762E11	(b)	.0296	.0068	2.6403+8	
69.10	.2292E10	.4762E11	(b)	.0481	.0068	2.6403+8	
69.10	.3174E10	.4762E11	(b)	.0667	.0068	2.6403+8	
36.04	(c)	.5695E11	.3911E8	.0009	.0001		Tennessee River
25.00	(c)	.2349E12	.4443E8	.0006	.0001		Ohio River
354.00	(c)	.4009E12	.4664E8	.0076	.0001		Mississippi River
273.00	(c)	.4071E12	.4670E8	.0058	.0001		
331.00	(c)	.4712E12	.4731E8	.0070	.0001		
255.00	(c)	.4994E12	.4753E8	.0054	.0001		
373.00	(c)	.5715E12	.4815E8	.0077	.0001		
				.3827 yr (140 days)			

[a] .4789E9 represents 4.789 x 10^8.

[b] Not required for reservoirs.

[c] Not required for rivers.

Table B7. Parameters of Missouri-Mississippi System

Length km	Volume m^3	Flow m^3/yr	Velocity m/yr	Transit Time, yr	Net Sedimentation Rate, m/yr	
60	(a)	.3063E11[b]	.3179E8	.0019	.0001	Missouri River
439	(a)	.3576E11	.3207E8	.0137	.0001	
352	(a)	.5000E11	.3280E8	.0107	.0001	
235	(a)	.7142E11	.3368E8	.0070	.0001	
105	(a)	.1727E12	.4324E8	.0024	.0001	Mississippi River
354	(a)	.4009E12	.4664E8	.0076	.0001	
273	(a)	.4071E12	.4670E8	.0058	.0001	
331	(a)	.4712E12	.4731E8	.0070	.0001	
255	(a)	.4994E12	.4753E8	.0054	.0001	
373	(a)	.5715E12	.4815E8	.0077	.0001	
				.0692 (25+ days)		

[a] Not applicable to river sections.

[b] .3063E11 represents 3.063 x 10^{10}.

B3.4.2.2 Distribution Coefficients

The appropriate values for the sediment distribution coefficients for a river system can vary significantly with the distance along the system. Inasmuch as these coefficients depend strongly upon the chemistry of the water, they can vary at any point at which the character of the water changes. For example, near the mouth of any significant tributary, the overall composition of the water can change enough to alter the distribution coefficients. Due to lack of sufficient information as to exactly what these changes should be, the values have been assumed to be constants for the entire lengths of the river systems. The values utilized in the calculations are the same as those employed for the Great Lakes.

B3.5 Series of Waterbodies

The complete impact of an accidental release of radioactivity into any waterbody is not necessarily determined by considering just the effect within that waterbody. The water from most waterbodies flows into other waterbodies. For example, the water from rivers often flows into estuaries, then into coastal reaches and finally into the deep portions of the ocean. The modeling of flow through the series of Great Lakes and down river systems has already been discussed.

For both the river and the estuary models, the fluxes of materials passing out through the mouths of these bodies have been calculated using

$$F_x = \iint \left(u_x c - D_x \frac{dc}{dx} \right) dy\ dz\ , \qquad \text{(B3-31)}$$

where u_x is the magnitude of the net velocity of the isotope under consideration, c is the concentration in the water and D_x is the longitudinal dispersion coefficient. When it was determined that a sufficiently large flux of any radionuclide had passed through the mouth of the river or estuary, then the efflux of that waterbody was used as input to the next waterbody in the series. No calculations were performed for impacts due to the residence of the radioactivity in the deep portions of the ocean.

B3.6 Limitations

All the modeling discussed in this appendix suffers from the typical shortcomings of any consideration of generic sites: the dominant features of a representative site are presumably included but the variations that exist among actual sites are not necessarily addressed or allowed for. Furthermore, the actual distributions of the required parameters are not known for most of the models. As a result, some of the parameters employed in the models might not be those which are most appropriate for a representative site. In addition, there may not even exist any genuinely representative sites.

B3.6.1 Limitations of Models

The purpose of this study is to consider the risk of exposure via liquid pathways relative to that through atmospheric pathways. The risk is determined on the basis of the radiation dose received by the population, along with the resulting health effects.

In a linear approximation (Appendix D), the resultant health effects are dependent only upon the total dose received by the population; they are not dependent upon the details of the distribution of that dose among the members of the population. Consequently, in modeling the population dose resulting from an accidental release, the use of transport models involving extensive averaging, such as the models used in this study, is often adequate.

In other health-effects models, such as the central estimate and the lower bound of the RSS (Appendix D), the resultant health effects depend not only upon the total population dose but also upon the doses received by individuals. The calculation of doses to individuals requires knowledge of the individual exposure or usage rates. Unless the distributions of the individual usage rates are adequately known, the use of more sophisticated water transport models than those used here will not necessarily improve the results.

Therefore, for this study, rather basic transport models have been employed. More detailed models should be used for site-specific studies and for proper calculation of doses to individuals.

All the surface-water transport models, except the one for river systems, employ simplifications such that analytic solutions are obtained. The simplifications include assumptions of homogeneous, isotropic mediums in which uniform flow or mixing occurs. The parameters for these models are taken as constants with respect to both spatial and temporal variations. In reality, the parameters are generally functions of both space and time. For example, the dispersion coefficients for all the models vary from point to point in any surface waterbody due to irregularities in the shoreline, among other factors. The dispersion coefficients also vary with time due to seasonal changes in the flow rates.

Unlike the other models, the river system model permits some variation of many of the parameters with respect to space and/or time. However, it does so in a rather limited fashion. In that model, the river system is divided into a series of homogeneous isotropic compartments, in each of which the parameters cannot vary. However, variation is permitted between the compartments. Although, in principle, the compartments can be defined to be as small as necessary to obtain the desired resolution, in practice, decreasing compartment size increases the numerical dispersion and consequently limits the attainable resolution. (In addition, sufficient data is rarely available to justify substantial resolution.)

All the sediment-water interaction models have assumed uniform rates of sedimentation (and resuspension) both spatially and temporally. Also, the sediment has been assumed to be the same in character throughout a model. In reality, the sedimentation (and resuspension) rates vary with both the location in the waterbody and the time.[39] In addition, the sediment in different locations is often of dissimilar composition. Furthermore, different processes can be responsible for removing various elements from the water column.[40] For example, one element might be removed by reactions with organic portions of the sediment while another might be removed primarily by interaction with the clay components of the sediment. Consequently, differing sedimentation rates for various sediment components could result in the removal of inequivalent amounts of certain elements as a function of either location or time.

In the models, adsorption of radionuclides onto sediment particles as the particles fall through the water has been assumed to be the primary removal mechanism for the nuclides. However, other significant removal mechanisms might exist such as seepage of surface water into groundwater or removal by aquatic organisms. Such processes have been ignored in the modeling.

B3.6.2 Limitations by Parameters

No models are better than the parameters used to exercise them. Only those parameters to which the final results are relatively insensitive can be ignored with any justification. However, not all the sensitive parameters (Section B4) are known to even within an order of magnitude accuracy.

It is beyond the scope of this study to determine exactly what a representative generic site is for most types of waterbodies; furthermore, it is also beyond the scope to estimate exactly what the applicable distribution of each of the significant parameters is. Consequently, it is not possible to determine adequately the error bounds which should be placed on the results.

A Great Lake site is the only type of site for which most of the required surface-water parameters can be obtained rather easily. Presumably these hydrological parameters could often be obtained for the other types of surface-water sites if sufficient effort were exerted. However, appropriate values for the distribution coefficients are generally not available for any type of surface waterbody for a wide range of radionuclides.

In contrast, the hydrological parameters required to describe groundwater travel are not available for many of the sites even if an effort is made. As in the surface-water case, the corresponding distribution coefficients are very rarely available with any level of effort. This study has at least attempted to determine what a representative site is with respect to the hydrological parameters describing groundwater travel from the reactor to the nearest surface waterbody of any consequence by surveying all the existing and proposed sites. Thus, an approximate distribution of groundwater transport parameters was obtained for all LWR sites.

In all the models, the parameters employed are treated as being completely independent. However, in actuality, the parameters can have some interdependencies. For example, the dispersion coefficients for the models should be a function of the speed of the water. Such relationships are ignored in this treatment except through intelligent choices of parameters for such coupled variables.

B4 Sensitivity Studies

B4.1 Introduction

In this report, the relevant consequences to humans are taken to be the doses of radiation received by the affected populations. For each liquid pathway, the dose obtained from steady-state dosimetry models is proportional to the concentration of activity in the water. (See Appendix D for specific expressions.) As a result, if one assumes that the population exposure rate for a given pathway does not vary with the passage of time, then the population dose via any liquid pathway is just proportional to the integral of the concentration over time and space of exposure. That is,

$$\text{population dose} \propto \int_0^{\infty} \int_{\substack{\text{area of}\\ \text{exposure}}} P(x,y) \cdot U \cdot c_i(x,y;t)\, dx\, dy\, dt. \tag{B4-1}$$

where P is the size of the exposed population per unit area and U is the individual usage rate. To consider the effects of the reasonable variation of certain parameters utilized in the water transport models, the values of the above concentration integral can be employed.

To compare the relative population doses for different waterbodies by using concentration integrals, one must include the appropriate population sizes or densities as weighting factors. If individuals can be assumed to have the same usage habits at various waterbodies, then the unweighted integrals can be compared directly to determine the relative average doses to individuals. That is,

$$\text{individual dose} \propto \int_0^{\infty} \int_{\substack{\text{area of}\\ \text{exposure}}} c_i(x,y;t)\, dx\, dy\, dt. \tag{B4-2}$$

This apparent proportionality of the concentration integral both to population dose and to individual dose is somewhat misleading.* Population doses can be calculated with average usage data and averaged concentrations. However, much more detailed models and data are probably required to realistically estimate actual individual doses than are needed to calculate population doses. Consequently, the expression for the individual dose given in Equation (B4-2) should be interpreted as being for an "average" individual unless more complicated concentration models than those of this study are used in it.

In the expressions for the dose, the integrations over time can be performed either for all time or for some finite period of interest. For isotopes which have short half-lives relative to the time period under consideration, the specified integration period is of little importance. However, for isotopes which have half-lives either comparable to or longer than the period of time under consideration, altering the relevant period of integration can significantly change the predicted results.

For most surface waterbodies, the flushing time is short compared to the time of interest in this study, that is, 100 years. Consequently, in surface waterbodies, the period of integration is generally effectively determined by the waterbody's flushing rate (and/or the decay rate) and not by the stated integration range. In contrast, the flushing time for aquifers is often much larger than 100 years. Therefore, the stated integration range can be important in interpreting the significance of the estimated dose, at least for long-lived radionuclides.

B4.2 Release Rate

To be effective in causing exposure via a pathway which involves a surface waterbody, a radionuclide must reach the waterbody before significant decay has taken place. For depressurization and sumpwater releases, this means that groundwater transport to the nearest surface waterbody must occur rapidly enough. For release by leaching of the melt debris, it means that contact of the groundwater with the melt, then leaching of the nuclide from the melt, and finally groundwater transport to the nearest surface waterbody must all occur rapidly enough.

The effects of varying the leaching rate alone can be investigated by considering the amount of a radionuclide which is leached before decay occurs. If one assumes a leaching rate equal to be^{-bt}, the amount of radioactive material leached before time t can be written as

$$\int_0^t b \exp[-(b+\lambda)t]\, dt = b(b+\lambda)^{-1} \{1 - \exp[-(b+\lambda)t]\}. \tag{B4-3}$$

---

*If the population is uniformly distributed, then

$$\text{population dose} \propto P \cdot U \int_0 \int_{\substack{\text{area of}\\ \text{exposure}}} c(x,y;t)\, dx\, dy\, dt$$

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The relative total amount leached for all time is presented in Table B8 as a function of both leach rate and half-life for some illustrative cases.

Leach rates are among the most important parameters of this study. They are also among the least well known.* (See Appendix A.) Obviously, the broad range of uncertainty connected with the leach rates makes less difference in the predicted final results for long-lived isotopes than for short-lived ones.

The total amount leached over all time is relatively independent of the specific expression employed for the leaching rate. For example, for a constant leach rate, the total amount released before time t can be written as

$$\int_0^t b\exp(-\lambda t)dt = (b/\lambda)\,[1-\exp(-\lambda t)]\,, \qquad \text{(B4-4)}$$

where the largest value of t is given by $t_b \equiv (1/b)$.

If one compares the results for a constant rate of release b and for the exponential expression assumed above, then one finds little difference in the total amounts leached prior to decay (Tables B8 and B9 and Figures B13 and 2.2). The total amount released at the constant rate always slightly exceeds the amount released exponentially (Figure 2.2). If one considers finite time periods, then the differences between the amounts leached according to the two different expressions varies somewhat more than the amounts for all time.

B4.3 Groundwater Movement

B4.3.1 Nondry Sites

Nondry sites are those at which exposure occurs only via a surface waterbody. The sole function of an aquifer at such a site is the transport of the radioactive material from the scene of the accident to the surface waterbody. The important quantity at such a site is the amount of radioactivity which crosses the aquifer-waterbody interface before decay occurs. Consequently, the integral of the concentration for the aquifer portion of the movement is not considered at these sites. Rather, the amount of material crossing the interface is determined so that it can be used as a weighting factor for the integrals obtained at various surface-water sites.

The total amount of material which crosses an interface at a distance x from the source and perpendicular to the direction of groundwater flow is given by the integral of the flux over time. For an instantaneous release from a point source into an aquifer, the amount which passes the interface before time t can be written as

$$\begin{aligned}\text{amount}(x) &= \int_0^t F_x dt \\ &= (-4)^{-1}(4D_x/a)^{-1/2}\exp(-\beta)\Big\{x\gamma^{-1/2}\,[\exp\{2(\alpha\gamma)^{1/2}\}\,\mathrm{erfc}\{(\gamma/t)^{1/2} \\ &\quad + (\alpha t)^{1/2}\} + \exp\{-2(\alpha\gamma)^{1/2}\}\,\mathrm{erfc}\{(\gamma/t)^{1/2} - (\alpha t)^{1/2}\} \\ &\quad - 2\exp\{-2(\alpha\gamma)^{1/2}\}] + ua^{-1}\alpha^{-1/2}\,[\exp\{2(\alpha\gamma)^{1/2}\}\,\mathrm{erfc}\{(\gamma/t)^{1/2} \\ &\quad + (\alpha t)^{1/2}\} + \exp\{-2(\alpha\gamma)^{1/2}\}\,\mathrm{erfc}\{-(\gamma/t)^{1/2} + (\alpha t)^{1/2}\} \\ &\quad - 2\exp\{-2(\alpha\gamma)^{1/2}\}]\Big\}\,, \qquad \text{(B4-5)}\end{aligned}$$

---

*If the composition and the character of the melt debris at a given site were known, the leach rates would still be highly uncertain. If one seeks to investigate the rates for a generic melt at a generic site, then the appropriate rates are even more uncertain. At the least, the possible rates vary over wide ranges.

Table B8. Relative Amount Leached as a Function of Leach Rate and Half-Life for a Leach Rate = be^{-bt} (a)

b (fraction/yr)→ Half-Life (yr)↓	10^{-4}	10^{-2}	10^{0}	10^{2}
1×10^{-2}	-	1.4×10^{-4}	1.4×10^{-2}	5.9×10^{-1}
1×10^{-1}	-	1.4×10^{-3}	1.3×10^{-1}	9.4×10^{-1}
1.0×10^{0}	-	1.4×10^{-2}	5.9×10^{-1}	9.9×10^{-1}
1.0×10^{1}	1.4×10^{-3}	1.3×10^{-1}	9.4×10^{-1}	1.0×10^{0}
1.0×10^{2}	1.4×10^{-2}	5.9×10^{-1}	1.9×10^{-1}	1.0×10^{0}
3.0×10^{1}	4.0×10^{-3}	3.0×10^{-1}	9.8×10^{-1}	1.0×10^{0}
3.0×10^{3}	3.0×10^{-1}	9.8×10^{-1}	1.0×10^{0}	1.0×10^{0}

[a] Relative amount = 1.0 implies that essentially the entire amount was leached prior to decay.

Table B9. Relative Amount Leached as a Function of Leach Rate and Half-Life for a Leach Rate = b[(a)]

b (fraction/yr)→ Half-Life (yr)↓	10^{-4}	10^{-2}	10^{0}	10^{2}
1×10^{-2}	1.4×10^{-6}	1.4×10^{-4}	1.4×10^{-2}	7.2×10^{-1}
1×10^{0}	1.4×10^{-4}	1.4×10^{-2}	7.2×10^{-1}	1.0×10^{0}
1×10^{2}	1.4×10^{-2}	7.2×10^{-1}	1.0×10^{0}	1.0×10^{0}
3.0×10^{1}	4.2×10^{-1}	3.8×10^{-1}	9.9×10^{-1}	1.0×10^{0}
3.0×10^{3}	3.8×10^{-1}	9.9×10^{-1}	1.0×10^{0}	1.0×10^{0}

[a] Relative amount = 1.0 implies that essentially the entire amount was leached prior to decay.

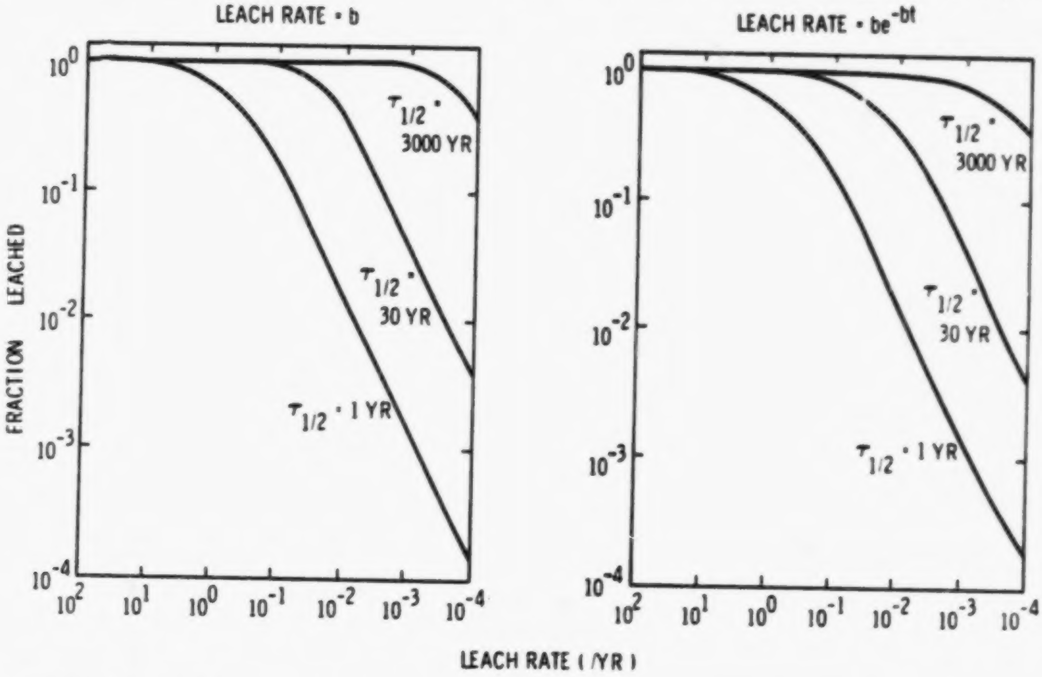

Figure B13. Fraction Released Prior to Decay for Two Different Leach Functions and for Various Leach Rates.

where

$$\alpha \equiv u^2/(4D_x a) + \lambda \ ,$$
$$\beta \equiv -xu/(2D_x) \ , \qquad \text{(B4-6)}$$
$$\gamma \equiv x^2 a/(4D_x) \ .$$

Similarly, for the same type of release, the total amount of material which passes the interface at x over all time is given by

$$\int_0^\infty F_x dt = (2)^{-1} (4D_x/a)^{-1/2} \exp[-\beta] \exp[-2(\alpha\gamma)^{1/2}]$$
$$\cdot \ \{x\gamma^{-1/2} + ua^{-1}\alpha^{-1/2}\} \ . \qquad \text{(B4-7)}$$

The fraction of the initial material which crosses an interface before decay is indicated in Tables B10 and B11 and in Figure B14 for three different radionuclides. Table B10 and Figure B14 consider that fraction as a function of both the average groundwater travel time, t_{H_2O}, and the distribution coefficient, K_d; Table B11 considers that fraction as a function of both t_{H_2O} and the hydrogeological dispersion.

As can be seen in both Table B10 and Figure B14, the fraction of the initial radioactivity which crosses the interface before decay is extremely dependent upon the radioactive half-life, the groundwater travel time, and the distribution coefficient. For a short-lived isotope, the fraction is negligible if either the travel time is relatively long or the distribution coefficient is relatively large. In contrast, for a very long-lived isotope, the fraction is small only if both of these factors are relatively large.

Some effects of varying the amount of dispersion are indicated in Table B11. Dispersion by the aquifer results in spreading of the radionuclide mass. The observed effect for liquid pathways is most significant for those cases in which the amount of radioactivity permitted to contact the human population is greatly changed, that is, when the amount of material entering a surface waterbody is substantially altered. For example, for an isotope with a relatively short half-life, a large dispersion can allow human contact with much more of the material before it decays than would otherwise be expected. However, if the water speed is low enough so that dispersion occurs in the upgradient direction and if the half-life is relatively short, then the net flux across the interface can be decreased by increasing the dispersivity; the material which disperses against the gradient can decay before it travels across the interface. In general, the net impact of increased dispersion is most important for relatively large values of both the distribution coefficient and the groundwater travel time and for relatively short radioactive half-lives.

As can be seen in Table B11, for an isotope with a half-life of 3000 years, with a K_d of 10^3 and with a groundwater travel time of 10 years (that is, with an average isotope travel time to the interface of approximately 50,000 years), almost 25 percent of the initial activity can cross the interface before decay has taken place, provided the dispersion is very large. However, if the dispersion is small, much less activity can cross the interface. If there is negligible dispersion, only 1.0×10^{-5} of the original amount crosses the interface before decay. The predicted resultant population doses for the two different dispersion coefficients would vary by a factor of 2.5×10^4.

Table B10. Fraction of Material Which Crosses a Plane at a Distance x Meters Downgradient From the Source After an Instantaneous Release into a Typical Aquifer.$^{(a)}$

	$\tau_{1/2}$ = 1 yr			$\tau_{1/2}$ = 30 yr			$\tau_{1/2}$ = 3000		
t_{H_2O} (yr)	10^{-1}	10^0	10^1	10^{-1}	10^0	10^1	10^{-1}	10^0	10^1
K_d									
0	9.33-1[b]	4.98-1	1.39-2	9.98-1	9.77-1	7.70-1	1.00+0	1.00+1	9.98-1
10	2.97-2	3.12-11	3.32-18	8.89-1	3.09-1	2.49-3	9.99-1	9.88-1	8.72-1
10^2	6.52-15	2.44-48	3.64-57	3.14-1	5.30-5	2.11-10	9.88-1	8.89-1	3.26-1
10^3	3.56-103	-	-	1.21-5	8.44-25	5.94-33	8.91-1	3.61-1	2.68-3
10^4	-	-	-	1.88-44	4.64-95	-	3.15-1	5.39-5	2.16-10

[a] t_{H_2O} represents the average travel time in years for the groundwater to go from the source to x. $D_x \simeq 50$ m²/yr.

[b] 9.33-1 denotes 9.33×10^{-1}.

Table B11. Fraction of Material Which Crosses an Interface at a Distance x Meters Downgradient From the Source After an Instantaneous Release into the Aquifer.$^{(a)}$

	$\tau_{1/2}$ = 1 yr			$\tau_{1/2}$ = 30 yr			$\tau_{1/2}$ = 3000 yr		
t_{H_2O} (yr)	10^{-1}	10^0	10^1	10^{-1}	10^0	10^{+1}	10^{-1}	10^0	10^1
$K_d = 0$									
$D_x = 50$	9.33-1	4.98-1	1.39-2	9.98-1	9.77-1	7.70-1	1.00+0	1.00+0	9.98-1
$D_x = 5000$	9.22-1	4.71-1	2.97-1	9.98-1	9.42-1	5.63-1	1.00+1	1.00+0	9.58-1
$K_d = 10^3$									
$D_x = 50$	-	-	-	1.21-5	8.44-25	5.94-33	8.91-1	3.16-1	2.68-3
$D_x = 5000$	-	-	-	2.68-3	3.30-4	2.57-4	8.74-1	3.79-1	2.48-1

[a] t_{H_2O} represents the average travel time in years for the groundwater to go from the source to x.

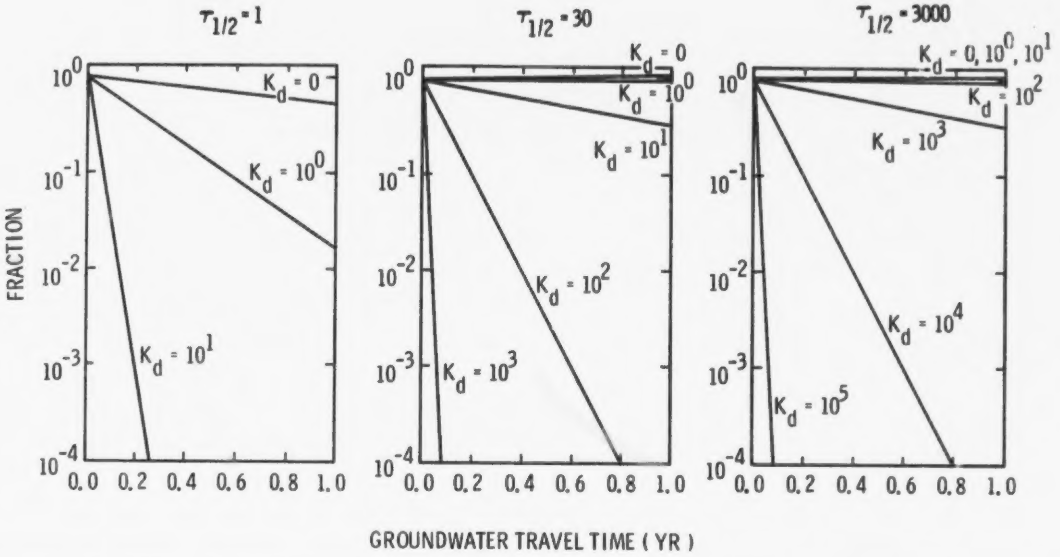

Figure B14. Fraction of Material Crossing Interface Before Decay as a Function of Groundwater Travel Time and Distribution Coefficient (K_d).

B4.3.2 Dry Sites

At a "dry" site, exposure occurs only through an aquifer. The previously-mentioned integral of the concentration, which is proportional to the population exposure, and not the flux is considered for dry sites.

Exposure via an aquifer will probably occur only beyond some finite distance downstream from the accident location. The integration over space from a distance x′ down the hydraulic gradient to infinity and the integration over all time of the radionuclide concentration due to an instantaneous release of material into an aquifer result in

$$I = \int_{-\infty}^{\infty}\int_{x'}^{\infty}\int_{0}^{\infty} c \; dt \; dx \; dy$$

$$= (2n_e ad)^{-1} (b + \lambda)^{-1/2} [(b + \lambda)^{1/2} - b^{1/2}]^{-1}$$

$$\cdot \; \exp[-x' \{(b + \lambda)^{1/2} - b^{1/2}\} (a/D_x)^{1/2}] \, , \qquad \text{(B4-8)}$$

where

$$b = u^2/(4D_x a) \; . \qquad \text{(B4-9)}$$

Although dispersion up the hydraulic gradient is usually not too important in an aquifer, it is straightforward to include such situations. The integration over space from a distance -x′ up the hydraulic gradient to infinity down the gradient and the integration over all time after an instantaneous release of material into an aquifer result in

$$I = \int_{-\infty}^{\infty}\int_{-x'}^{\infty}\int_{0}^{\infty} c(x,y;t) \; dt \; dx \; dy$$

$$= (2n_e ad)^{-1} (b + \lambda)^{-1/2} \{[(b + \lambda)^{1/2} - b^{1/2}]^{-1}$$

$$- [(b + \lambda)^{1/2} + b^{1/2}]^{-1} (\exp[-x' \{(b + \lambda)^{1/2} + b^{1/2}\} (a/D_x)^{1/2}] - 1)\} \, .$$

$$\text{(B4-10)}$$

If the dispersion coefficient is very large or if the groundwater velocity is very low, then the second major term in the outer braces can be significant. In the extreme situation in which the groundwater speed is zero, the dispersion dominates and the first term in the outer braces (the downgradient contribution) is equal to the second term (the upgradient contribution), provided -x′ is taken far enough away from the source.

Rather than considering the integration over all time, one might just integrate over a finite period to determine the results caused in that time period. Such integrations have also been considered for all the spatial variations just mentioned, using

$$\int_0^t t^{-1/2} e^{-(at+b+c/t)}dt = \exp(-b)\,(\pi/a)^{1/2}(-2)^{-1}$$

$$\{\exp[2(ac)^{1/2}]\,\mathrm{erfc}[(c/t)^{1/2}+(at)^{1/2}]$$

$$+\exp[-2(ac)^{1/2}]\,\mathrm{erfc}[-(c/t)^{1/2}+(at)^{1/2}]\}\,, \qquad \text{(B4-11)}$$

and

$$\int e^{fx}\,\mathrm{erfc}(gx+h)dx = \exp(-fh/g)\,(f)^{-1}$$

$$\cdot\ \{\exp[fz/g] - \exp[fz/g]\,\mathrm{erf}(z) + \exp[f^2/(4g^2)]$$

$$\mathrm{erf}[z-(f/2g)] + \text{constant}\,, \qquad \text{(B4-12)}$$

where

$$z \equiv gx + h\ . \qquad \text{(B4-13)}$$

The concentration integrals for the examples considered in Table B12 and Figure B15 are strongly dependent upon both the distribution coefficient and the half-life. For each of the half-lives discussed, the integrals vary approximately inversely as the values of the K_d's. As mentioned previously, the ratio of the amount of a nuclide retained by the soil to the amount in solution varies directly as K_d. Consequently, an increased K_d just means that a larger amount of the nuclide resides on the soil and is therefore unavailable for contact with humans.

Larger values of K_d always result in a more significant upstream contribution to the integral. This is because the larger values of K_d correspond to the greater delaying of the isotopes relative to the groundwater movement; therefore, more dispersion can occur upstream from the source.

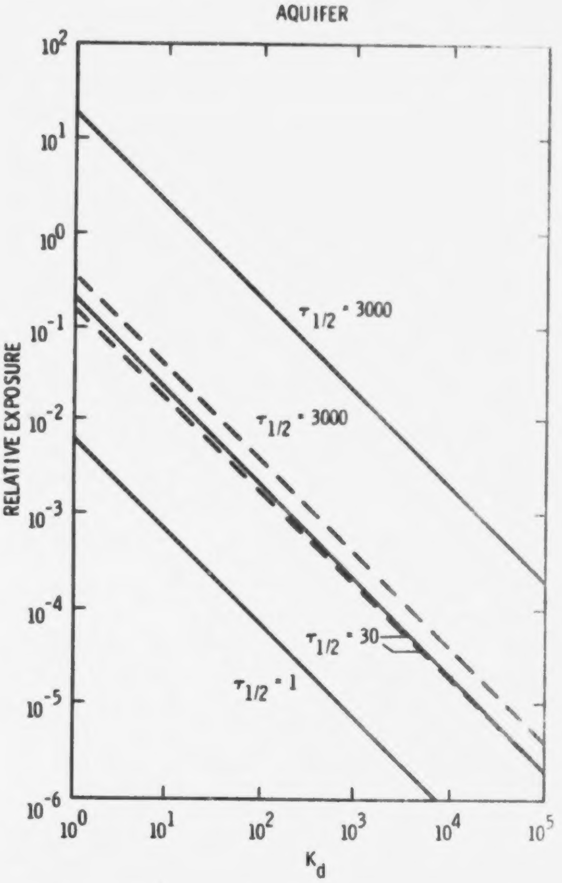

Figure B15. Concentration Integral for an Instantaneous Release into an Aquifer Versus Distribution Coefficient. (Solid lines represent integrals over all time; dashed lines represent integrals from t = 0 to t = 100 years.)

Table B12. Concentration Integrals Over All Time and Over the First 100 Years After an Instantaneous Release into an Aquifer of Infinite Extent.

	0 to Infinity			0 to 100 years	
	$\tau_{1/2} = 1$	$\tau_{1/2} = 30$	$\tau_{1/2} = 3000$	$\tau_{1/2} = 30$	$\tau_{1/2} = 3000$
Standard	7.21-6	2.16-4	2.16-2	1.92-4	3.21-4
No delay	3.60-2	1.08+0	1.02+2	9.73-1	2.32+0
$10^2 \times K_d$	7.21-8	2.16-6	2.16-4	1.95-6	4.54-6
$10 \times K_d$	7.21-7	2.16-5	2.16-3	1.93-5	3.89-5
$K_d/10$	7.20-5	2.16-3	2.16-1	1.90-3	3.43-3
$K_d/10^2$	7.07-4	2.12-2	2.12+0	1.88-2	4.37-2

Standard case: $K_d = 10^3$; $n_e = 0.2'$; d = 200 m; u = 50 m/yr; $D_x = 500\ m^2/yr$; K_d is the distribution coefficient; n_e is the effective porosity; d is the depth of the aquifer; u is the speed of the water; D_x is the longitudinal dispersion coefficient.

Because the aquifer of the dry site has been taken to be of infinite extent and because the entire area of the source has been integrated over, the concentration integrals in Table B12 and in Figure B15 are independent of the groundwater velocity. If a finite area of integration were employed instead, the concentration integrals would be extremely dependent upon the velocity. As expected from the analytical expression for the concentration integral, the integral is inversely proportional to the depth, the width and the effective porosity of the aquifer.

Two periods, one consisting of the 100 years following the accident, as well as one of infinite temporal extent, have been considered in Table B12 and Figure B15. As can be seen, the period of integration is not very important for a half-life of either 1 or 30 years; however, for a 3000 year half-life, changing the integration period from 100 years to all time increases the value of the concentration integral by approximately two orders of magnitude. Consequently, the period of integration employed for an aquifer must be specified for isotopes with long half-lives.

B4.4 Surface Waterbody Movement

B4.4.1 Lake

Exposure via a lake can occur both in the immediate vicinity of the accident and at some distance from the site. The magnitude of the population dose due to exposure close to the accident relative to that farther away depends upon both the relative sizes of the exposed populations and the relative amounts of dilution in the two areas.

In the nearshore region, the integration over space from a distance x down-current from the accident site to a distance x′ up-current and the integration over all time of the radionuclide concentration resulting from an instantaneous release of material into the lake produce

$$I = (y)^{-1} \int_0^y \int_{-x}^{x'} \int_0^\infty C \, dt \, dx \, dy \, , \qquad \text{(B4-14)}$$

where C is given by Equations (B3-2) and (B3-3) for $\leq 8.3 \times 10^{-3}$ years (3 days) and by Equation (B3-16) for $t > 8.3 \times 10^{-3}$ years.

Typical values for the contribution to the integral of the first three days are listed in Table B13. Due to the relatively rapid flushing of the nearshore region, the integrals are approximately the same for a wide range of half-lives. Except for very short-lived isotopes, the integral over the nearshore zone for the first three days is negligible compared to the integral over the entire lake for all time. Consequently, the integral over the nearshore zone will not be considered further. The uniformly mixed-tank model will be utilized for all further sensitivity studies for the Great Lakes.

The major portion of a Great Lake is described in terms of a complete mixing model. In such a model, the concentration in the lake is independent of the location in the lake. Therefore, the spatial integrations over the lake are trivial. As a result,

$$I = \int_0^\infty C \, dt. \qquad \text{(B4-15)}$$

Using the expression for the concentration in a completely mixed lake with no sediment effects, Equation (B3-5), results in an integral over all time that can be written as

$$I = (Q/V) \left[(q/V) + \lambda \right]^{-1} . \qquad \text{(B4-16)}$$

Similarly, the concentration integral for a finite interval of time can be written as

$$I = (Q/V) \left[(q/V) + \lambda \right]^{-1} \left[1 - \exp\left\{ - \left[(q/V) + \lambda \right] t \right\} \right] . \qquad \text{(B4-17)}$$

As might be expected, the integral for a completely mixed lake is inversely proportional to the volume of the lake. Also as expected, if either decay or flushing occurs rather rapidly, then the integral will be comparatively small. If either the flushing rate for the lake or the radioactive decay rate for the radionuclide is significantly larger than the other rate, then the integral is also inversely proportional to the larger rate (Figure B16). In other words, the faster process dominates the integral. As a result, for an isotope with a relatively long half-life, the integral is just approximately proportional to the inverse of the flow rate. That is,

$$I \simeq Q \, q^{-1} \, . \qquad \text{(B4-18)}$$

As can be seen in Figure B16, this results in the concentration integrals for all relatively long-lived radionuclides being essentially equal in lakes with comparatively rapid flushing rates.

Table B13. Concentration Integrals for an Instantaneous Release into the Nearshore Region of a Lake.[a]

$\tau_{1/2}$(yr) u(cm/sec)	2.19-2	1.0-1	1.0+0	3.0+1	3.0+4
10	7.03-7	7.74-7	7.93-7	7.95-7	7.95-7
20	7.03-7	7.74-7	7.94-7	7.95-7	7.95-7

[a] A 3-day period is considered. The water depth is 10m; the nearshore region width is 10^3m.

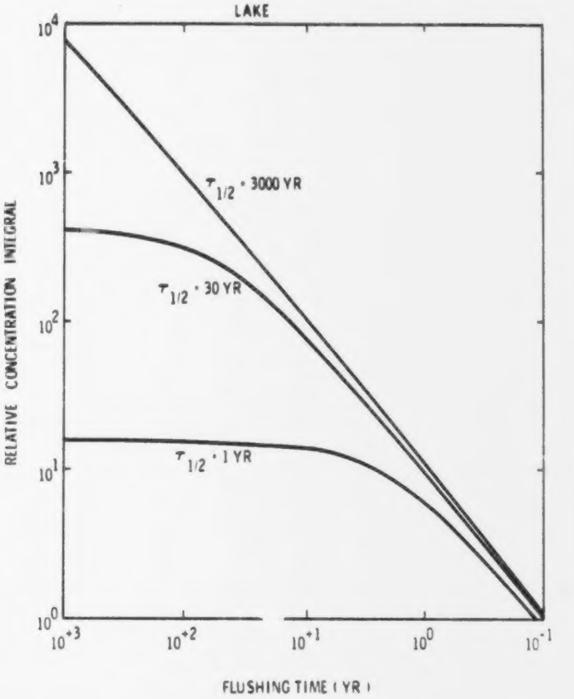

Figure B16. Relative Concentration Integrals for a Lake Site as a Function of Flushing Time. (Sediment interactions are not included.)

All of the Great Lakes except Lake Erie experience complete turnover twice a year. The effects of this seasonal turnover on the concentration integrals for two lakes are presented for a specific example in Table B14. In the example, complete turnover is assumed to occur six months after the release. Turnover is taken to be important only for the lake into which the release occurs.

Table B14. Concentration Integrals for Instantaneous Releases Into Lakes With and Without Seasonal Turnover.[a]

	No Turnover	Turnover
Ontario		
$\tau_{1/2}$ = .1	8.64-14	4.23-13
= 1.	7.44-13	1.74-12
= 30.	4.08-12	5.26-12
= 3000.	4.82-12	6.00-12
Michigan		
$\tau_{1/2}$ = .1	2.95-14	1.44-13
= 1.	2.83-13	7.28-13
= 30.	3.71-12	4.12-12
= 3000.	6.32-12	6.73-12

[a] Turnover is taken to first occur at 6 months after the accident. No turnover occurs in Lake Erie.

The net result of including turnover is an increase of the relative importance of the more rapidly decaying isotopes. The relative effect is more noticeable for the shorter lived isotopes; however, the absolute magnitude of the effect is greater for the longer lived ones. The concentration integrals for some isotopes increase by as much as a factor of five. The changes in all of the concentration integrals are more noticeable in lakes with greater flushing rates; in Lake Ontario, the net effect of including turnover is to increase the population dose due to each radionuclide by at least 20 percent, whereas, in Lake Michigan, the increase is only 5 to 10 percent.

In the example presented in Table B14, turnover was taken to occur when it would result in the maximum impact. If turnover occurred soon after the accident, it would lessen the magnitude of the calculated impact.

For the mth lake in a series of lakes in all of which sediment effects are neglected, the integration of the radionuclide concentration over a finite time period after an instantaneous release into the first lake produces

$$I_m = \frac{Q \prod_{\ell=1}^{m-1} q_\ell}{\prod_{\ell=1}^{m} V_\ell} \cdot \frac{\sum_{j=1}^{m} \{\exp(-\lambda_{ij}t) - 1\}}{\prod_{k=1,\neq j}^{m+1} [\lambda_{ik} - \lambda_{ij}]} , \qquad \text{(B4-19)}$$

where

$$\lambda_{ij} \equiv (q_j/V_j) + \lambda_i \qquad \text{for } j \leq m,$$

$$\lambda_{ij} \equiv 0 \qquad \text{otherwise.} \qquad \text{(B4-20)}$$

$\{I_j\}$ can be used directly to compare the relative doses to individuals with the same usage habits on different lakes. However, to compare the relative population doses on different lakes, the appropriate populations sizes must be used as weighting factors.

As can be seen in Table B15, the concentration integrals for lakes after the first in a series can be significant compared to those of the first lake for longer-lived isotopes. The differences between the concentration integrals over all time and over the first 100 years after the accident tend to be greatest as one moves down the series, that is, for the last lakes reached.

The previous considerations have not included sediment-nuclide interactions. For an instantaneous release into a lake in which sediment-nuclide interactions are included in the manner described in Subsection B3.1,

$$I = [V(S_1 - S_2)]^{-1} \cdot \{(\lambda_4 + S_1)(S_1)^{-1} [\exp(S_1 t) - 1]$$

$$- (\lambda_4 + S_2)(S_2)^{-1} [\exp(S_2 t) - 1]\} , \qquad \text{(B4-21)}$$

where S_1 and S_2 are as defined in Equations (B3-17) and λ_4 is as defined in Equation (B3-15).

For relatively high values of the distribution coefficient, this expression can be written approximately as

$$I \approx V^{-1} (\lambda_2)^{-1} [1 - \exp(-\lambda_2 t)] , \qquad \text{(B4-22)}$$

where λ_2 is as described in Equation (B3-15). That is,

$$\lambda_2 \approx (q/V) + \lambda + (vK_d/d_w) . \qquad \text{(B4-23)}$$

For relatively low values of the distribution coefficient, the same approximation for the concentration integral holds except that

$$\lambda_2 \approx (q/V) + \lambda . \qquad \text{(B4-24)}$$

Table B15. Concentration Integrals for Lakes in a Series for Various Half-Lives ($\tau_{1/2}$ = 1 yr, 30 yr, 3000 yr)[a]

	t Ranges From 0 to Infinity			t Ranges From 0 to 100 Years	
	$\tau_{1/2} = 1$	$\tau_{1/2} = 30$	$\tau_{1/2} = 3000$	$\tau_{1/2} = 30$	$\tau_{1/2} = 3000$
Michigan	2.83-13	3.71-12	6.32-12	3.70-12	6.08-12
Huron	1.59-14	2.38-12	6.29-12	2.34-12	5.52-12
Erie	5.10-15	2.04-12	5.70-12	1.99-12	4.95-12
Ontario	6.57-16	1.44-12	4.76-12	1.39-12	3.96-12
Huron	3.58-13	4.09-12	6.33-12	4.08-12	6.23-12
Erie	1.15-13	3.49-13	5.74-12	3.49-12	5.64-12
Ontario	1.48-14	2.47-12	4.79-12	2.46-12	4.67-12
Erie	2.04-12	5.45-12	5.78-12	5.45-12	5.78-12
Ontario	2.63-13	3.85-12	4.82-12	3.85-12	4.82-12
Ontario	7.44-13	4.08-12	4.82-12	4.08-12	4.82-12

[a] The first mentioned lake in each group is the lake into which a unit release is assumed to have occurred. Sediment interactions are ignored.

Substantial sediment effects are not observed in a single Great Lake for isotopes with distribution coefficients less than 10^3 (Table B16 and Figure B17). The relatively low net rates of sedimentation found in all these lakes result in a slow rate of sediment interaction. Consequently, K_d must be comparatively large for a significant amount of interaction to occur. As expected, changing the sedimentation rate for a relatively large K_d has a very noticeable effect on the results whereas changing the sedimentation rate for a comparatively small value of K_d makes little difference in the results. Furthermore, the sediment effects are greater in the lakes with the longer residence times; the slower flushing rates in these lakes permit more time for interactions to occur.

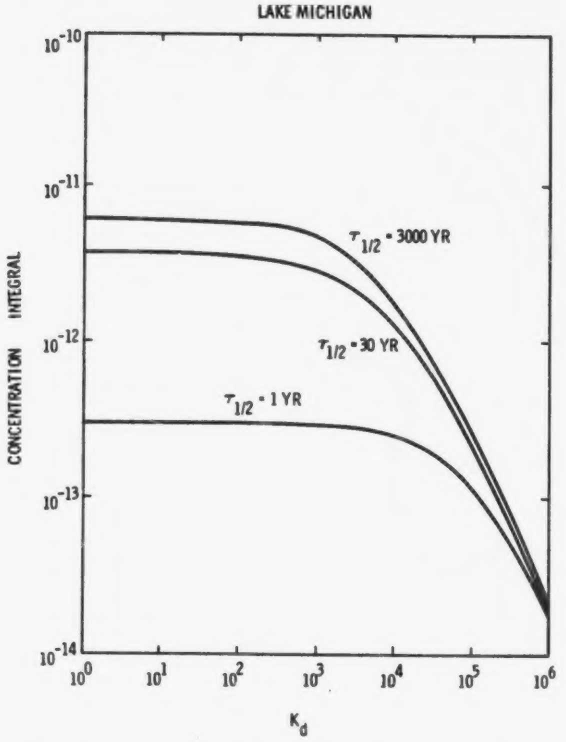

Figure B17. Concentration Integrals for an Instantaneous Release into Lake Michigan Versus Sediment Distribution Coefficient.

The effects of sediment interactions are greater for long-lived isotopes. This is because short-lived isotopes decay before they can undergo significant interaction. In the limit of relatively large K_d ($>10^5$), sediment effects dominate the picture and result in approximately equal concentration integrals for the half-lives considered here (Figure B17).

As can be seen in Table B17, the cumulative effect of sediment-nuclide interactions is much more striking as one progresses through a series of lakes than the effect is for just a single lake. This is especially true for high values of K_d. Consequently, the effects of lakes after the first should be included for relatively low values of the distribution coefficient but not necessarily for high values. The larger effect in a series is due to the longer effective residence time for the radionuclide in the series.

The flushing of Lake Michigan may occur more slowly than indicated by the 30.8 year residence time given in Table B4.[*] A longer residence time in Lake Michigan would tend to increase the estimated dose received from that lake. For relatively short-lived isotopes, little change would be observed in the population dose; however, for comparatively long-lived nuclides, the change in population dose could be significant due to the comparatively large populations associated with Lake Michigan. For a 100-year residence time and a half-life of 30 years, the population dose received from Lake Michigan could be increased by as much as 70 percent; for a half-life of 3000 years, the dose could be increased by over 200 percent. For a 30 year half-life, the population doses in succeeding lakes would be much smaller than those calculated for the 30.8 year residence time; however, for the 3000 year half-life, the doses in other lakes would be essentially unchanged.

B4.4.2 Estuary

Exposure via an estuary can occur both upstream and downstream from the location of an accidental release. The extent to which exposure occurs upstream depends upon the importance of the tidal mixing relative to that of net seaward flow.

The integration over space from $-x_u$ upstream above the source to x_d downstream and the integration over all time of the radionuclide concentration resulting after an instantaneous release of material into the estuary produce

$$I = (y)^{-1} \int_{-\infty}^{\infty} \int_{-x_u}^{x_d} \int_{0}^{\infty} c \, dt \, dx \, dy \, . \qquad \text{(B4-25)}$$

---

[*] The overall flushing pattern for Lake Michigan is more complex than that for any of the other three lakes. Lakes Huron, Erie, and Ontario are characterized essentially by continuous flow-through; Lake Michigan, however, is not. The 30.8 year residence time for Lake Michigan is considered by some to be an approximate minimum time.

That is,

$$I = (2Aa)^{-1}(b + \lambda)^{-1/2}\{[b^{1/2} - (b + \lambda)^{1/2}]^{-1}(\exp x_d\{b^{1/2}$$

$$- (b + \lambda)^{1/2}\}/E_L'^{1/2}] - 1) - [b^{1/2} + (b + \lambda)^{1/2}]^{-1}(\exp[-x_u b^{1/2}]$$

$$+ (b + \lambda)^{1/2}\}/E_L'^{1/2}] - 1\}, \qquad \text{(B4-26)}$$

where

$$b \equiv u'^2/(4E_L'). \qquad \text{(B4-27)}$$

c for this integration was taken from Equation (B3-22).

Table B16. Concentration Integrals Over All Time for an Instantaneous Release Into the Lake Considered With Sediment Effects Taken Into Account. ($\tau_{1/2}$ = 1 yr, 30 yr, 3000 yr.)

	Michigan			Erie			Ontario		
	$\tau_{1/2} = 1$	$\tau_{1/2} = 30$	$\tau_{1/2} = 3000$	$\tau_{1/2} = 1$	$\tau_{1/2} = 30$	$\tau_{1/2} = 3000$	$\tau_{1/2} = 1$	$\tau_{1/2} = 30$	$\tau_{1/2} = 3000$
Standard	2.76-13	2.89-12	4.81-12	2.01-12	5.25-12	5.64-12	7.29-13	3.70-12	4.47-12
No sediment	2.83-13	3.71-12	6.32-12	2.04-12	5.45-12	5.78-12	7.44-13	4.08-12	4.82-12
$10^2 \times K_d$	1.18-13	1.93-13	1.99-13	1.05-12	1.55-12	1.58-12	3.34-13	5.28-13	5.41-13
$10 \times K_d$	2.46-13	1.27-12	1.55-12	1.85-12	4.31-12	4.57-12	6.58-13	2.39-12	2.69-12
$K_d/10$	2.80-13	3.47-12	6.12-12	2.02-12	5.41-12	5.77-12	7.37-13	3.99-12	4.79-12
$K_d/10^2$	2.81-13	3.67-12	6.29-12	2.03-12	5.45-12	5.79-12	7.40-13	4.08-12	4.83-12
$4 \times K_f$	2.68-13	2.50-12	4.79-12	1.97-12	5.12-12	5.63-12	7.09-13	3.47-12	4.46-12
$K_f/4$	2.79-13	3.07-12	4.82-12	2.02-12	5.30-12	5.64-12	7.34-13	3.79-12	4.47-12
$2 \times v$	2.73-13	2.55-12	3.90-12	1.99-12	5.14-12	5.49-12	7.20-13	3.50-12	4.17-12
$v/2$	2.78-13	3.08-12	5.43-12	2.02-12	5.31-12	5.71-12	7.33-13	3.80-12	4.64-12
$5 \times d_s$	2.76-13	2.82-12	4.73-12	2.01-12	5.23-12	5.63-12	7.29-13	3.66-12	4.45-12
$d_s/5$	2.76-13	3.01-12	4.82-12	2.01-12	5.29-12	5.64-12	7.29-13	3.77-12	4.47-12
$3 \times d_w$	2.82-13	3.51-12	5.94-12	2.03-12	5.41-12	5.76-12	7.41-13	4.00-12	4.75-12
$d_w/3$	2.52-13	1.53-12	2.46-12	1.88-12	4.56-12	5.10-12	6.71-13	2.68-12	3.37-12

Standard case: $K_d = 10^3$; $K_f = .4$ m/yr; $v = 5 \times 10^{-4}$ m/yr; $d_s = .1$ m; $d_w = 50$ m; K_d is the distribution coefficient; K_f is the diffusion rate; v is the average sedimentation rate; d_s is the depth of the interacting sediment; d_w is the depth of the water column.

Table B17. Comparison of Concentration Integrals With and Without Inclusion of Sediment Interactions for All Time for an Instantaneous Release Into a Lake in a Series.[a]

	$\tau_{1/2} = 1$			$\tau_{1/2} = 30$			$\tau_{1/2} = 3000$		
	$K_d=0$	$K_d=10^3$	$K_d=10^4$	$K_d=0$	$K_d=10^3$	$K_d=10^4$	$K_d=0$	$K_d=10^3$	$K_d=10^4$
Michigan	2.83-13	2.79-13	2.58-13	3.71-12	3.05-12	1.61-12	6.32-12	4.62-12	1.97-12
Huron	1.59-14	1.54-14	1.29-14	2.38-12	1.59-12	4.31-13	6.29-12	3.37-12	6.17-13
Erie	5.10-15	4.66-15	3.09-15	2.04-12	1.18-12	1.94-13	5.70-12	2.64-12	2.87-13
Ontario	6.57-16	5.92-16	3.71-16	1.44-12	7.81-13	9.80-14	4.76-12	2.04-12	1.62-13
Erie	2.04-12	2.02-12	1.91-12	5.45-12	3.12-12	9.92-13	5.78-12	3.22-12	1.00-12
Ontario	2.63-13	2.59-13	2.42-13	3.85-12	1.65-12	2.33-13	4.82-12	1.93-12	2.48-13
Ontario	7.44-13	7.35-13	6.94-13	4.08-12	3.83-12	2.91-12	4.82-12	4.47-12	3.27-12

[a] Sediment effects are included approximately. ($v = 5 \times 10^{-4}$ m/yr; $K_f = 0.4$ m/yr; d_w for each lake is the appropriate average depth given in Table B4).

Except for a multiplicative factor dependent upon the rate of release, this result for an instantaneous release is identical to that obtained for steady-state conditions resulting from a continuous release of constant magnitude. The first major term in the outer braces represents the contribution from downstream, while the second is the contribution from upstream. If the net flow of water is relatively large, then the upstream portion of the result is expected to be rather small. However, in the limit of no net flow, the upstream and the downstream contributions become equal. It is easily seen that for nonzero net flow, the upstream contribution is always less than the downstream contribution.

Burial can represent a significant removal mechanism in an estuary. If decay is negligible, then the removal of radioactive flux can be used as an indicator of the amount of that burial. The amount of material which passes through the mouth of an estuary after an instantaneous release from a point source can be written as

$$\begin{aligned}\text{amount} &= \int_o^t f_x dt \\ &= -\frac{1}{4\sqrt{4E_L'}} e^{-\beta} \Big\langle \frac{x}{\gamma^{1/2}} \Big[e^{2(\alpha\gamma)^{1/2}} \operatorname{erfc}\Big\{\Big(\frac{\gamma}{t}\Big)^{1/2} + (\alpha t)^{1/2}\Big\} \\ &\quad + e^{-2(\alpha\gamma)^{1/2}} \operatorname{erfc}\Big\{\Big(\frac{\gamma}{t}\Big)^{1/2} - (\alpha t)^{1/2}\Big\} - 2e^{-2(\alpha\gamma)^{1/2}}\Big] \\ &\quad + \frac{u}{a\alpha^{1/2}} \Big[e^{2(\alpha\gamma)^{1/2}} \operatorname{erfc}\Big\{\Big(\frac{\gamma}{t}\Big)^{1/2} + (\alpha t)^{1/2}\Big\} \\ &\quad + e^{-2(\alpha\gamma)^{1/2}} \operatorname{erfc}\Big\{-\Big(\frac{\gamma}{t}\Big)^{1/2} + (\alpha t)^{1/2}\Big\} - 2e^{-2(\alpha\gamma)^{1/2}}\Big]\Big\rangle ,\end{aligned} \tag{B4-28}$$

where

$$\begin{aligned}\alpha &\equiv u'^2/(4E_L') + \lambda , \\ \beta &\equiv -xu'/(2E_L') , \\ \gamma &\equiv x^2/(4E_L') .\end{aligned} \tag{B4-29}$$

Concentration integrals for an estuary are shown in Table B18 and Figures B18 and B19 and the fractions of radioactive material which pass out of an estuary prior to decay are shown in Table B19 and Figure B20 for some sample cases. The results both with and without burial are included. (The comparison of the two sets of results permits evaluation of the effect of the burial model.)

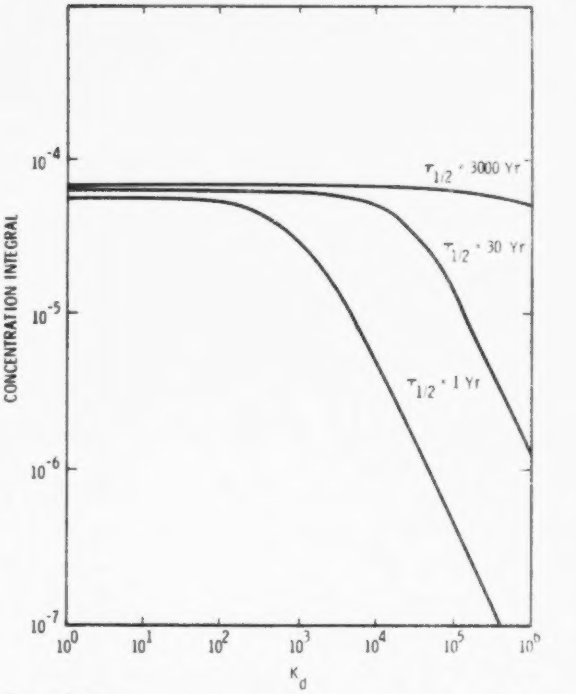

Figure B18. Concentration Integral for an Instantaneous Release Into an Estuary Without Burial Versus Sediment Distribution Coefficient.

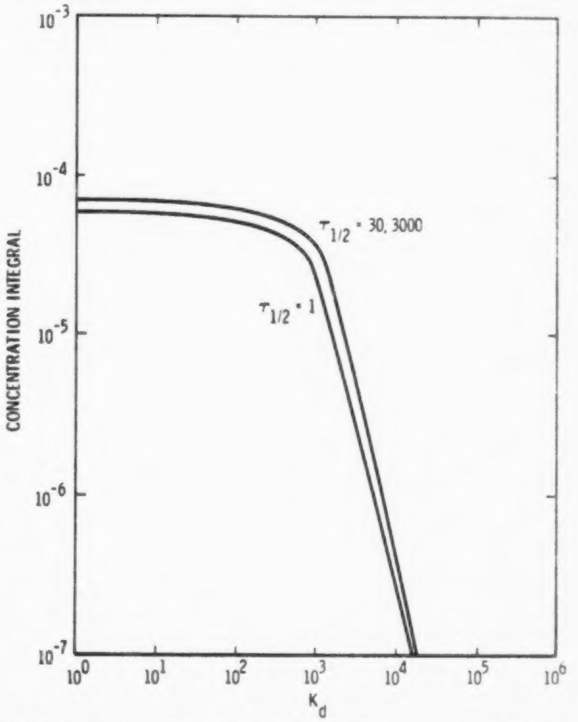

Figure B19. Concentration Integral for an Instantaneous Release Into an Estuary With Burial Versus Sediment Distribution Coefficient.

Table B18. Concentration Integrals for an Instantaneous Release Into an Estuary With Sediment Effects Taken Into Account ($\tau_{1/2}$ = 1 yr, 30 yr, 3000 yr.)

	With Burial			Without Burial		
	$\tau_{1/2} = 1$	$\tau_{1/2} = 30$	$\tau_{1/2} = 3000$	$\tau_{1/2} = 1$	$\tau_{1/2} = 30$	$\tau_{1/2} = 3000$
Standard	9.25-6	1.65-5	1.69-5	1.53-5	3.19-5	3.29-5
No sediment	2.94-5	3.25-5	3.29-5	2.94-5	3.25-5	3.29-5
$10^2 \times K_d$	1.43-9	1.45-9	1.45-9	2.17-7	6.45-6	3.19-5
$10 \times K_d$	1.53-7	1.74-7	1.75-7	2.15-6	2.54-5	3.28-5
$K_d/10$	2.54-5	3.12-5	3.14-5	2.71-5	3.27-5	3.29-5
$K_d/10^2$	2.80-5	3.16-5	3.18-5	2.90-5	3.27-5	3.29-5
$2 \times v$	6.60-6	1.03-5	1.05-5	1.53-5	3.19-5	3.29-5
$v/2$	1.44-5	2.20-5	2.27-5	1.53-5	3.19-5	3.29-5
$5 \times d_s$	1.98-6	4.71-6	4.93-6	4.34-6	2.95-5	3.37-5
$d_s/5$	2.00-5	2.54-5	2.56-5	2.50-5	3.24-5	3.27-5
$3 \times d_w$	2.02-5	2.83-5	2.87-5	2.29-5	3.23-5	3.27-5
$d_w/3$	1.68-6	2.47-6	2.51-6	6.90-6	3.07-5	3.33-5
$10 \times u$	2.75-6	2.98-6	2.99-6	2.94-6	3.19-6	3.20-6
$u/10$	6.20-6	1.45-5	1.51-5	1.89-5	2.58-4	4.13-4
$10 \times E$	6.60-6	1.21-5	1.25-5	1.45-5	3.91-5	4.15-5
$E/10$	1.01-5	1.74-5	1.78-5	1.54-5	3.10-5	3.20-5
$\ell \times 2$	9.90-6	2.10-5	2.18-5	-	-	-

Standard case: $K_d = 10^3$, $v = 8 \times 10^{-3}$ m/yr; $d_s = .1$ m; $d_w = 15$ m; $u = 3.16 \times 10^5$ m/yr; $E = 10^9$ m^2/yr; $A = 10^4$ m^2; length = ± 100 km.

Table B19. Fraction of Material Which Passes Through the Mouth of an Estuary After a Unit Instantaneous Release ($\tau_{1/2}$ = 1 yr, 30 yr, 3000 yr.)

	Without Burial			With Burial		
	$\tau_{1/2} = 1$	$\tau_{1/2} = 30$	$\tau_{1/2} = 3000$	$\tau_{1/2} = 1$	$\tau_{1/2} = 30$	$\tau_{1/2} = 3000$
Standard	6.00-2	2.63-1	2.77-1	1.93-1	9.44-1	9.99-1
No Sediment	7.98-1	9.92-1	1.00+0	7.98-1	9.92-1	1.00+0
$10^2 \times K_d$	0.00+0	0.00+0	0.00+0	2.06-24	1.22-2	9.43-1
$10 \times K_d$	1.22-22	4.26-21	4.83-21	1.32-5	6.05-1	9.95-1
$K_d/10$	6.70-1	9.59-1	9.71-1	6.89-1	9.88-1	1.00+0
$K_d/10^2$	7.86-1	9.90-1	9.98-1	7.87-1	9.92-1	1.00+0
$2 \times v$	2.00-2	8.03-2	8.44-2	1.93-1	9.44-1	9.99-1
$v/2$	1.06-1	4.92-1	5.20-1	1.93-1	9.44-1	9.99-1
$5 \times d_s$	3.33-5	4.88-3	5.91-3	1.61-3	7.74-1	9.97-1
$d_s/5$	4.04-1	6.58-1	6.70-1	5.95-1	9.83-1	1.00+0
$3 \times d_w$	4.11-1	8.11-1	8.30-1	4.90-1	9.76-1	1.00+0
$d_w/3$	1.04-5	1.68-4	1.86-4	1.52-2	8.54-1	9.98-1
$10 \times u$	7.42-1	8.73-1	8.78-1	8.45-1	9.94-1	1.00+0
$u/10$	1.53-4	3.62-3	4.15-3	1.70-3	5.43-1	9.93-1
$10 \times E$	1.12-1	2.90-1	3.01-1	2.32-1	9.31-1	9.99-1
$E/10$	5.20-2	2.60-1	2.75-1	1.87-1	9.45-1	9.99-1
$\ell \times 2$	3.89-3	7.20-2	7.99-2	3.89-2	8.93-1	9.99-1

Standard case: $K_d = 10^3$, $v = 8 \times 10^{-3}$ m/yr; $d_s = .1$ m; $d_w = 15$ m; $u = 3.16 \times 10^5$ m/yr; $E = 10^9$m^2/yr; $A = 10^4$ m^2; length = ± 100 km.

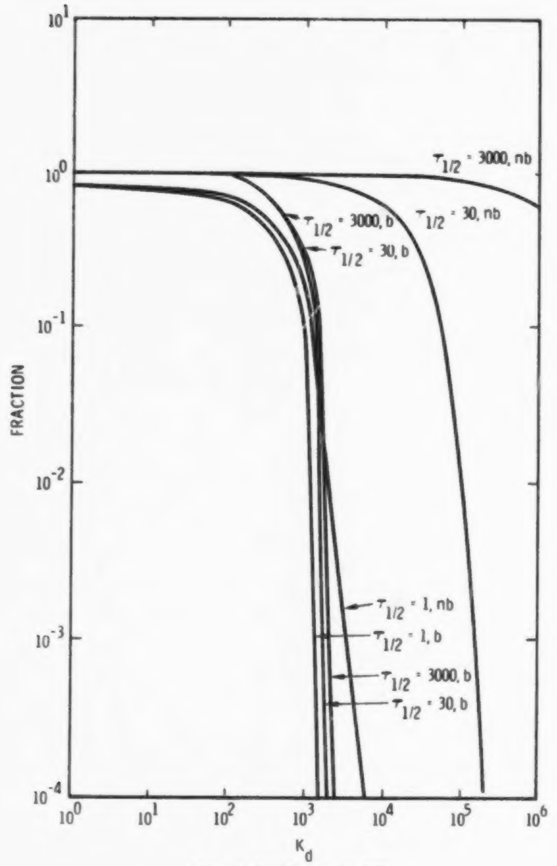

Figure B20. Amount Leaving an Estuary After an Instantaneous Release Versus Sediment Distribution Coefficient.

As K_d increases, the effects of sediment-water interactions become more apparent; more of the radioisotopes are removed from the water column by the sediments. Therefore, both the integral over the concentration and the amount of radioactivity that flows out of the estuary decrease as K_d increases.

Increasing the sediment depth results in little change if K_d is small; however, if K_d is relatively large, a significant effect can be observed with change in that depth. The larger the amount of sediment is that is assumed to interact with water, the greater the activity is which can be effectively removed from the water column. Consequently, increasing the sediment depth generally decreases both the concentration integral and the amount of radioactivity flowing out of the estuary.

As the tidal dispersion increases, the upstream contribution to the concentration integral increases while the downstream portion decreases. The net effect is essentially no change in either the total concentration integral, or the flux out of the estuary.

Increasing the net speed of the water always results in a larger amount of material being carried farther downstream. However, the effect upstream depends upon the value of the distribution coefficient. If K_d is large, a greater flux is also observed upstream; if K_d is small, a smaller flux results upstream.

For the parameters used in the standard cases, in Tables B18 and B19, the effective net speed downstream of a nuclide ($K_d \approx 10^3$) is just over one-eighth that of the water. As a result, whereas the water takes approximately 0.3 years to move from the point considered as the source to the downstream extremum of the estuary, a standard-case nuclide takes about 2.5 years to move the same distance.

For a standard case isotope with a half-life of 30 years (and $K_d = 10^3$), approximately $e^{-.0575}$ or 94 percent of the initial activity passes out of the estuary prior to decay if burial is ignored. For the same isotope in an estuary with a net water speed which is slower by a factor of 10, approximately $e^{-.575}$ or only 56 percent of the activity leaves the estuary prior to decay. If burial is included, then substantially less activity passes out of the mouth of the estuary in both cases.

The behavior of the concentration integral for an estuary with burial with respect to the distribution coefficient (Figure B19) is somewhat different than that observed for either an estuary without burial (Figure B18) or a lake (Figure B17). Because of both the relatively short residence time of water in an estuary and the equilibrium assumption for the sediment-nuclide interactions for the upper layer of sediment, the isotopes considered have essentially the same concentration integrals for any given value of K_d if burial is included. In contrast, if burial is not included, then the concentration integrals for radionuclides with longer half-lives are larger for a given K_d. For high values of K_d, much decay of shorter lived isotopes can occur in the estuary while those isotopes are residing on the sediments. Consequently, for large K_d's, the concentration integrals are relatively small.

Figure B20 depicts the total amount of undecayed material leaving an estuary versus the sediment distribution coefficient. Cases both with and without burial are given. Without burial, the amount is very dependent upon both the radioactive half-life and the distribution coefficient. With burial, the amount is essentially independent of the half-life for the considered values; however, the amount is extremely dependent upon the distribution coefficient. The amount escaping through the mouth starts to fall rapidly for $K_d > 10^3$; for large values, very little radioactivity escapes from the standard-case estuary.

Figure B21, which plots the relative value of the concentration integral versus flushing time for an estuary, is quite similar to that for a lake (Figure B16). The major difference is the time at which the isotopes become essentially equivalent.

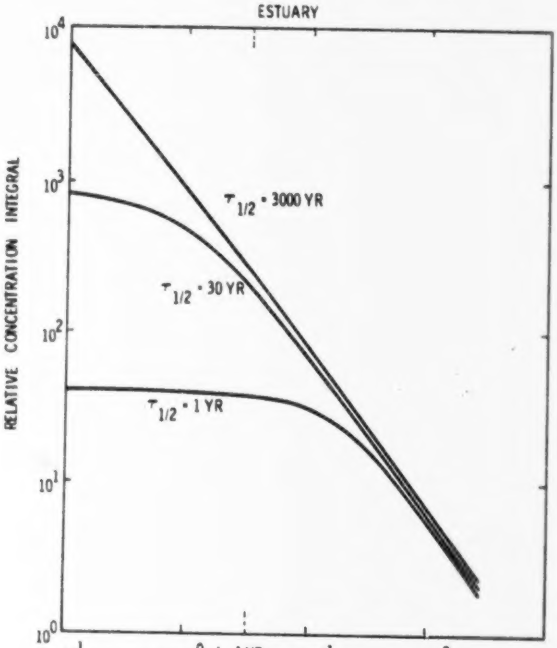

Figure B21. Concentration Integral for an Instantaneous Release Into an Estuary as a Function of Flushing Time. ($K_d = 10^3$)

B4.4.3 Ocean

Exposure via an ocean can occur both in the immediate vicinity of the accident and at some distance from the site. The magnitude of the population dose due to exposure close to the accident relative to that farther away depends both on the relative sizes of the exposed populations and on the relative dilution which occurs.

The integration over space from $-x_u$ upcurrent above the source to x_d downcurrent and the integration over all time of the radionuclide concentration resulting from an instantaneous release of material into a narrow sheltered tidal bay produce

$$I = (y)^{-1} \int_0^y \int_{-x_u}^{x_d} \int_0^\infty C \, dt \, dx \, dy \, . \qquad \text{(B4-30)}$$

That is,

$$I = (2dy)^{-1} (b + \lambda)^{-1/2} \{[b^{1/2} - (b + \lambda)^{1/2}]^{-1} (\exp[x_d\{b^{1/2} - (b + \lambda)^{1/2}\}/E_L^{1/2}] - 1) - [b^{1/2} + (b + \lambda)^{1/2}]^{-1} (\exp[-x_u \{b^{1/2} + (b + \lambda)^{1/2}\}/E_L^{1/2}] - 1)\} , \qquad \text{(B4-31)}$$

where

$$b^2 = u^2/4E_L \, . \qquad \text{(B4-32)}$$

Similarly, the integration over space from $-x_u$ to x_d and over all time after an instantaneous release into the nearshore region of an open coastal region produces essentially the same integral as that listed above; however, the appropriate input parameters will differ greatly from those for the sheltered bay situation. If one integrates over a large enough spatial interval, then

$$I \approx (2dy)^{-1} (b + \lambda)^{-1/2} \{[(b + \lambda)^{1/2} - b^{1/2}]^{-1} + [(b + \lambda)^{1/2} + b^{1/2}]^{-1}\} . \qquad \text{(B4-33)}$$

The concentration integrals for the nearshore region of an open ocean site (Table B20) are very sensitive to the magnitude of the water velocity and to the water depth assumed. They are not very sensitive to the dispersion. These integrals are dominated by the relatively rapid flow and dilution which are characteristic of the nearshore regions of the ocean. This results in making the integrals for this region almost equal for a very wide range of half-lives.

The concentration in the offshore region is independent of the location. Consequently, the concentration integral for that region reduces to

$$I = \int_0^t C \, dt \, . \qquad \text{(B4-34)}$$

The exact form of this integral was given in Equation (B4-17).

Some concentration integrals for the offshore region are presented in Table B21. The concentration integrals for the offshore region are very sensitive to both the flushing rate (Figure B22) and the volume. For large flushing rates, the integrals are comparable for a wide range of half-lives (Figure B22). Consequently, if a rapid release occurs into a relatively rapidly flushed region of one of the continental

Table B20. Area Concentration Integrals for the Nearshore Region of an Ocean After an Instantaneous Release into the Region

	t = 0 to 0.1 yr					t = 0 to 10^2 yrs			
$\tau_{1/2}$ (yr):	2.19-2	1.0-1	1.0+0	3.0+1	3.0+3	1.0-1	1.0+0	3.0+1	3.0+3
10^2 km									
Standard	4.02-7	5.70-7	6.30-7	6.35-7	6.35-7	5.70-7	6.30-7	6.35-7	6.35-7
10 x u	6.05-8	6.30-8	6.35-8	6.35-8	6.35-8	6.30-8	6.35-8	6.35-8	6.35-8
u/10	6.05-7	1.44-6	1.93-6	2.00-6	2.00-6	2.57-6	5.70-6	6.35-6	6.35-6
10^3 km									
Standard	6.05-7	1.44-6	1.93-6	2.00-6	2.00-6	2.57-6	5.70-6	6.35-6	6.35-6
10 x u	4.02-7	5.70-7	6.30-7	6.35-7	6.35-7	5.70-7	6.30-7	6.35-7	6.35-7
u/10	6.05-7	1.44-6	1.93-6	2.00-6	2.00-6	2.84-6	2.57-5	6.15-5	6.35-5

Standard Case: $u = 3.15 \times 10^6$ m/yr; $E = 3.15 \times 10^6$ m^2/yr; $d_w = 10$ m; width $= 5 \times 10^3$ m.

shelves, then isotopes with a rather wide range of half-lives must be considered. However, if the release takes place after some delay, then the isotopes with shorter half-lives need not be dealt with at an ocean site.

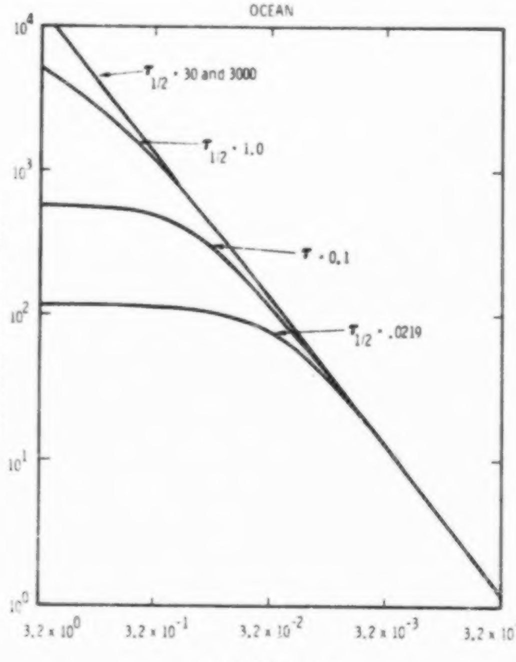

Figure B22. Relative Concentration Integral for an Ocean Site as a Function of Flushing Time.

Table B21. Concentration Integrals for the Offshore Region After an Instantaneous Release Into the Nearshore Region.

$\tau_{1/2}$(yr)→ Residence Time↓ (yr)	1.0-1	1.0+0	3.0+1	3.0+3
Standard (.5)	6.73-14	4.19-13	9.56-13	1.00-12
4	6.96-14	5.30-13	1.83-12	2.00-12
1	6.31-14	2.95-13	4.89-13	5.00-13

Standard case: volume $= 2 \times 10^{12}$ m^3; $(q/V) = .5$/yr.

At certain ocean sites, the initial release would be into a relatively sheltered embayment. This would tend to decrease the relative importance of the isotopes with shorter half-lives. However, it would tend to increase the net impact if sufficient pathways existed in the embayment itself.

B4.4.4 River

The form of the river system model does not permit integration over the concentrations to be performed analytically; that integration must be done numerically. As a result, much less extensive sensitivity work was performed for the river system model than was done for any of the other models.

For both of the river systems considered in this study, the rate of release and the timing of the release relative to flood peaks were investigated. The rate of release was varied by considering 1 Ci each of Sr-90, Cs-137, and Ru-106, either as 100 Ci/yr for 0.01 yr, or as 1 Ci/yr for 1 yr. No significant difference was found in the concentration integrals for the different release rates, although the concentration itself as a function of time was, of course, greatly affected.

The effects of seasonal flow variation were tested by making instantaneous releases at different times in the annual flow cycle. The concentration integrals for a 1 Ci instantaneous release of Cs-137 into the upper end of the MM system are given in Table B22. For this system, the concentration integrals vary at most by a factor less than three and generally by less than a factor of two even though flow rates vary by a factor of seven. The concentration integrals for the TOM system also do not vary substantially as a function of the release date.

The last line in Table B22 shows the results with constant average flow. The maxima in the runs for seasonally variable flow are slightly over a factor of two larger than these and the minimal are less than a factor of two smaller. The seasonally averaged values are much less than a factor of two higher than the values using average flow.

Table B22. Effect of Date of Instantaneous Release on Concentration Integrals for the Missouri-Mississippi System

Date of Release	$\int\int c dx dt$	$\int\int c B dx dt$[a]
March 1	2.4-5	1.0-2
June 1	3.5-5	1.3-2
Sept 1	6.7-5	3.4-2
Dec 1	3.7-5	1.5-2
Average	4.1-5	1.8-2
Constant Average Flow	2.8-5	1.2-2

[a] Proportional to the population dose from recreational use if usage is taken to be proportional to the surface area; B is the width of the river and is taken to be a function of x.

Table B23 shows the concentration integrals for unit rapid (0.01 yr) releases of Ru-106, Sr-90, and Cs-137 into the upper ends of both systems. The concentration integrals vary significantly with isotope for the TOM system but not for the MM system. Most of the indicated differences are due to the effects of sediment-nuclide interactions.

For steady releases into both systems, the water concentrations for all the nuclides tested arrived very nearly at steady values within 2 years. Sediment concentrations for Ru-106 also arrived at steady values within 2 years. However, the sediment concentrations for Sr-90 and Cs-137 did not arrive at steady values at the river mouth within 25 years; nearly constant values were achieved for these nuclides in at least the upper reaches. This indicates that a steady-state solution is adequate for water concentrations and for sediment contaminations in the upper reaches, but is often not adequate for calculating sediment concentrations throughout the lengths of such long river systems.

Figure B23 shows water concentrations for steady unit releases into the two river systems being considered. Part A of Figure B23 shows concentrations for the TOM system. The differences between the nondecaying contaminant and Ru-106 are principally due to decay of ruthenium during the travel time. The calculated differences between Sr-90 and Cs-137 are primarily due to difference in the assumed distribution coefficients.

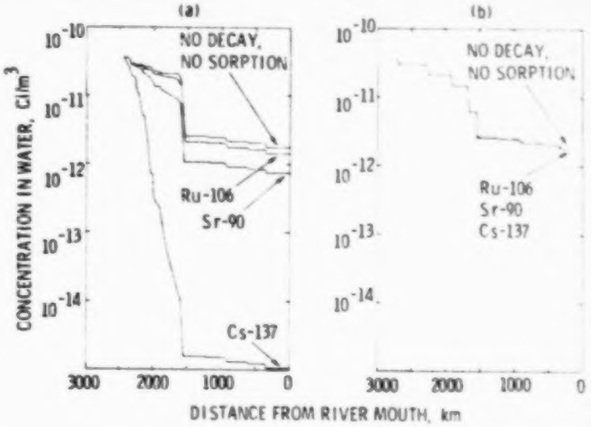

Figure B23. Steady-State Water Concentrations as a Function of Distance from the Mouth of the River for: (a) the Tennessee-Ohio-Mississippi System and (b) the Missouri-Mississippi System. (The values assumed for the distribution coefficient in these calculations are 1 for Ru, 2700 for Sr and 27000 for Cs.)

In contrast to the case for the water concentrations, steady-state sediment concentrations were not achieved by Sr-90 or Cs-137. Figure B24 shows the maximum sediment concentrations 25 years after the start of release. The discontinuities in the depicted sediment concentration profiles are artifacts of the model parameters; they are due to the assumption of two short sedimentless stretches in the TOM system.

Part B of Figure B23 shows water concentrations in the MM system for continuous releases. Sediment concentrations are not shown for this system. Because the travel time is so short and the sediment effects so small, the traces for all three nuclides are too close to be distinguishable.

Table B23. Concentration Integrals for Instantaneous Unit Releases into River Systems

River System	Nuclide	Kd	$t_{1/2}$(yr)	$\int\int$cdxdt	$\int\int\int$cBdxdt[a]
Tennessee-Ohio-Mississippi	Ru-106	1	1.005	2.2-4	4.9-2
	Sr-90	2,400	28.1	1.7-5	3.2-2
	Cs-137	27,000	30.23	5.2-6	7.7-3
Missouri-Mississippi	Ru-106	1	1.005	2.8-5	1.2-2
	Sr-90	2,400	28.1	2.8-5	1.2-2
	Cs-137	27,000	30.23	2.8-5	1.2-2

[a] Proportional to the population dose from recreational use if usage is proportional to the surface area.

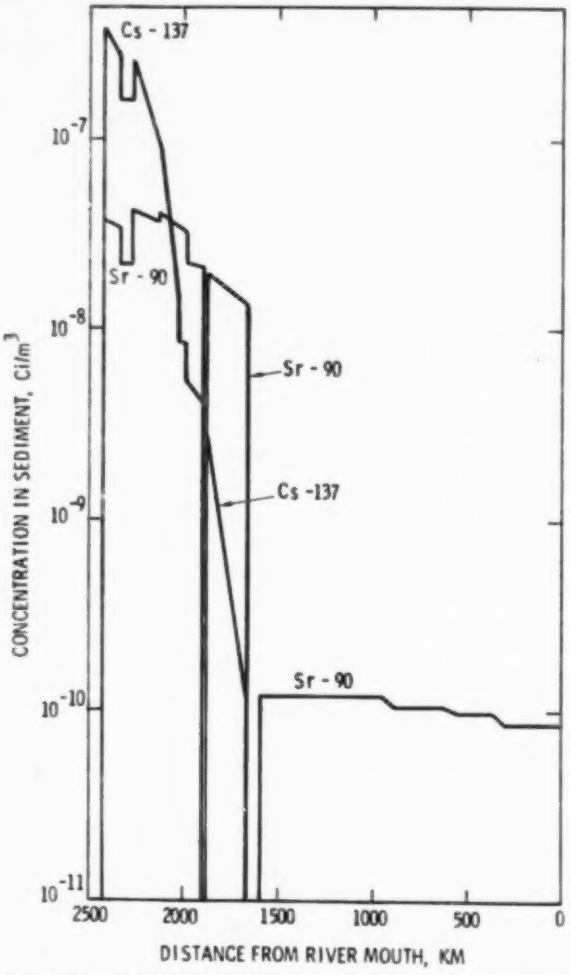

Figure B24. Maximum Sediment Concentrations Within 25 years of the Initiation of Release as a Function of Distance From the River Mouth for the Tennessee-Ohio-Mississippi System.

The transient response of each system is depicted in Figures B25 and B26. These figures show the variation in the water and sediment concentrations at the river mouth in response to instantaneous injections of 1 Ci at the upper ends of the systems. Figure B25 displays the water concentrations in the two systems after the release of Ru-106. Part A of that figure depicts the relatively dispersive character of a heavily dammed river (TOM). Part B of that figure reflects the comparatively nondispersive nature of a free-flowing river (MM).

Without dispersion, sorption, and decay, both systems would have given a pulse at the mouth with a concentration of 1.75×10^{-10} Ci/m^3 and a pulse width of 0.01 years. The flattening of the pulse in Part A of Figure B25 is almost solely due to dispersion; thus, fluvial dispersion alone reduces the peak concentration by more than a factor of two.

Figure B26 shows the transient response of the TOM system to the instantaneous release of 1 Ci of Cs-137. The concentration in the water at the mouth is down three orders of magnitude from the peak of Part A of Figure B25. The tail, beginning at 0.6 years, represents diffusion of Cs back into the water from the sediment and continues for several years with little diminution.

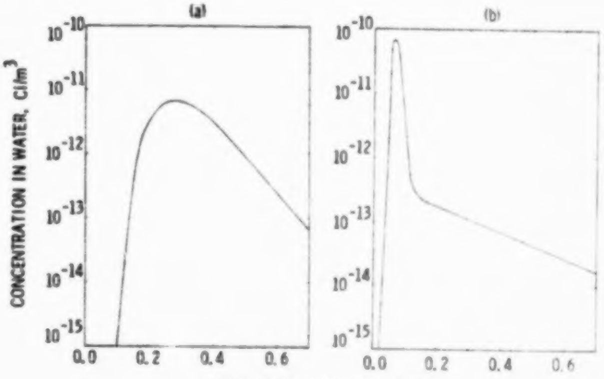

Figure B25. Water Concentration at the River Mouth in Response to Injection of 1 Ci of Ru-106 into (a) the Tennessee-Ohio-Mississippi System and (b) the Missouri-Mississippi System.

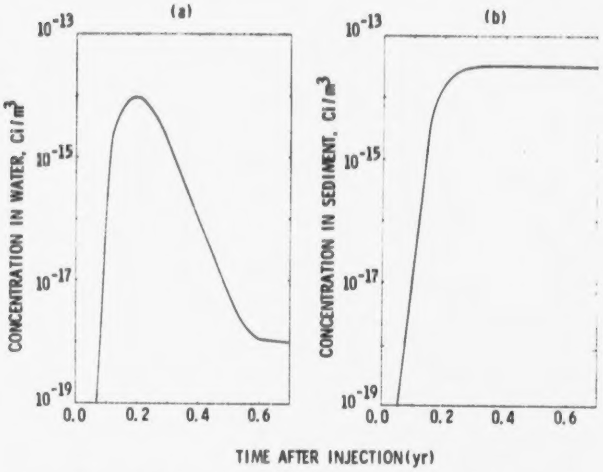

Figure B26. Transient Response of the Tennessee-Ohio-Mississippi System in Response to the Injection of 1 Ci of Cs-137: (a) Water Concentration and (b) Sediment Concentration.

Part B of Figure B26 shows the transient response of sediment concentration in that system for an instantaneous injection of Cs-137. The peak sediment concentration is about a factor of three higher than the water concentration and decays extremely slowly.

B4.4.5 Summary of Water Transport Sensitivities

The parameters for which the models discussed in the previous subsections are most sensitive are listed in Table B24. For all but the aquifer at a dry site, the flushing time of the waterbody dominates the sensitivity. The dry site exception is an artifact of the particular "dry site" aquifers discussed in this study. For direct human exposure, this report considers only aquifers of infinite longitudinal and lateral extents; consequently, the dry site aquifers discussed all have infinite flushing times.

Parameters such as depth and width in aquifers and estuaries have essentially simple inverse relationships to the concentration integrals. Therefore, even though variations of these parameters result in large changes in the integrals, the effects of these alterations are easily approximated. The relationships of the other parameters cannot usually be as readily estimated.

For aquifers, the integrals are extremely sensitive to any change in the distribution coefficient. The integrals for the isotopes considered are approximately inversely proportional to the value of K_d. For lakes, estuaries and rivers, the integrals are sensitive to the distribution coefficient and to other sediment-related properties only for large enough values of K_d. In all these waterbodies, the results are rather insensitive to a wide variation in K_d for relatively small and moderate values.

For a large lake, that is, for a Great Lake, sediment effects are not very important for values of the sediment distribution coefficient which are less than approximately 10^3. Even for values larger than this, the reductions are only significant for isotopes with comparatively long half-lives. This is because of the relatively slow net sedimentation rate, and hence the slow rate of sediment-nuclide interactions, in the Great Lakes. The effect of sediment interactions is greater for a lake in a series than for an isolated lake; passage through a series of lakes allows more time for sediment-water interactions to occur.

For an estuary, the range of the sediment distribution coefficient over which the results are rather insensitive depends strongly on whether burial is included in the model. If burial is considered, the results are very sensitive for values of $K_d > 10^2$. Unlike the situation in other surface waterbodies, the integrals for an estuary are very sensitive to certain changes in the value of the longitudinal dispersion coefficient.

For a river system, the importance of sediment effects depends, in part, upon the residence time of the water in the system. In a system with reservoirs, the residence time of the water tends to be relatively long and the sedimentation rate tends to be high. For example, the average residence time for the Tennessee-Ohio-Mississippi system is approximately 140 days. In that system, the results are sensitive only to values of $K_d > 10^4$. In contrast, in a free-flowing system, the residence time of the water tends to be short and the sedimentation rate tends to be low. The average residence time for one such system, the Missouri-Mississippi system, is approximately 25 days. In that system, the results are sensitive only to values of $K_d > 10^6$ or more.

Only the concentration integral for an aquifer is extremely sensitive to the period of integration; all the other waterbodies considered have their effective periods of integration determined by their flushing times. For all except the aquifer, these are usually less than 100 years. For certain waterbodies other than the ones considered here, this is not always the situation.

Table B24. Sensitive Parameters (other than Half-Lives) for Concentration Integrals

Waterbody	Very Sensitive	Somewhat Sensitive	Sensitive but Straightforward
Aquifer	Flushing rate (u_x;ℓ); K_d;D_x	--	d;y;n_e
Lake	Flushing rate	K_d[a]; other sediment-related parameters	Volume
Estuary	Flushing rate (u_x, ℓ)	K_d[b]; other sediment-related parameters; E_x	d,y
Ocean	Flushing rate	--	Volume
River	Flushing rate (u_x, ℓ)	K_d[c]; other sediment-related parameters	$\{d\}$, $\{y\}$

[a] Especially sensitive to variations in K_d and related sediment parameters if $K_d > 10^3$.

[b] Especially sensitive to variations in K_d and related sediment parameters if $K_d > 10^2$.

[c] Especially sensitive to variations in K_d and related sediment parameters if $K_d > 10^4$ for a river with several reservoirs and if $K_d > 10^6$ for a river without reservoirs.

B4.5 Combined Sensitivity Studies

B4.5.1 Sensitivities in General

In general, for any release of radioactivity into a portion of the hydrosphere, there are two primary phenomena on which the description is dependent: the hydrological flushing rate and the strength of the soil-nuclide or sediment-nuclide interactions. The lower the flushing rate of a waterbody is, the longer the residence time of both the radionuclides and the water are in that waterbody. If the residence is in a portion of the hydrosphere with no significant exposure pathways to the human population, then a slow flushing rate is preferable. However, if the residence is in a portion including one or more significant pathways, then a rapid flushing rate is best. The stronger the soil-nuclide or sediment-nuclide interactions in a waterbody are, then the lower the resultant concentrations of the radionuclides in the water are. Consequently, for all but exposure related to contaminated sediments, stronger interactions tend to result in a lower exposure of humans.

Prolonging a release into a waterbody tends to decrease the rate at which the exposure is obtained but it does not necessarily substantially decrease the total amount of exposure. It diminishes the total amount only if the protraction permits significant decay to occur before exposure.

B4.5.2 Sensitivities in Aquifers

The description of the transport of radioactivity through aquifers is extremely dependent on any variations in either the flushing rate of the aquifer or the strength of the soil-nuclide interactions. If exposure is through wells in the aquifer, then a slow flushing rate for the aquifer can increase the impact on the human population. However, if exposure is through surface water, then a slow flushing rate, and therefore a long groundwater travel time, for the aquifer can decrease the impact by permitting substantial decay to occur before the surface water pathways are reached.

If the distribution coefficients are large, then the travel of the nuclides through the aquifer will be relatively slow. In such a situation two primary factors can lower the impact on the human population: substantial decay can occur during the delayed transport; and the concentration in the water can be drastically lowered by "absorption" (for example, see Figure B15). The effects of prolonging a release into an aquifer are expected to be most noticeable when the prolongation permits decay to reduce the amount of radioactivity which reaches the population.

B4.5.3 Sensitivities in Surface Waterbodies

For any type of surface waterbody, the population exposure is rather sensitive to any reasonable variations in the flushing rates; in contrast to the aquifer situation, the net exposure is sensitive only to certain limited variations in the strength of the sediment-nuclide interactions within the surface waterbody. The sensitivities to the sediment-nuclide interactions for any radionuclide are typically characterized by two portions: a rather broad plateau over which the population exposure is not especially sensitive to variations in any sediment parameters; and a region in which the population exposure is relatively sensitive to any alterations in certain sediment-related parameters (for example, see Figure B17). The plateau of insensitivity for any surface waterbody is broadest in situations in which the sediment interactions are slowest, that is, in situations in which the sedimentation rate and the resuspension rate are both low. (In general, as the ratio of the amount of interacting sediment to the quantity of available water increases, the sediment interactions become relatively more important. In the limit of approximately equal amounts of sediment and water, that is, in an aquifer, the results are extremely sensitive to any change in the sediment (soil) interaction paramenters.)

If the flushing rate is low, as for a Great Lake, then after an instantaneous release the amount of radioactivity in the water decreases relatively slowly with time. Consequently, the population exposure resulting from long-lived radionuclides can increase substantially with time (for example, see Figure B16). However, if the flushing rate is high, as for a free-flowing river, then after an instantaneous release the amount of radioactivity in the water decreases rapidly with time. As a result, the population exposure does not increase much with time (for example, see Figure B16). Furthermore, all the radionuclides tend to be essentially equivalent in systems with high flushing rates.*

B4.5.4 Availability and Importance of Data

For any given body of surface water, the hydrological turnover parameters can be approximated rather accurately. This is, in part, due to the rather narrow range of reasonable values for some of these parameters. Consequently, although the sensitivity to these parameters is important, it is not critical in greatly limiting the accuracy of the results. The appropriate sediment-water interaction parameters are not nearly as readily determined. However, the results are not very sensitive to these except in situations of very large sediment-water interactions. As a result, choice of parameters in the insensitive plateau region can usually be justified.

For an aquifer, the hydrological parameters are not as easily determined as they are for a surface waterbody. This is partly due to the relatively wide range of reasonable values for some of these parameters. With sufficient effort, reasonable approximations for the hydrological parameters can be obtained for a given site. In contrast, the soil-nuclide interaction parameters are not as easily determined. Furthermore, these parameters are very site-specific and cannot be easily generalized. Orders of magnitude of variation can exist between parameters appropriate for different sites; large variation can even exist at different locations of a given site. Unlike the surface waterbody situation, the results are extremely sensitive to any variation in the soil-nuclide interaction parameters. Consequently, the changes resulting from reasonable variations in these parameters must be kept in mind. Inasmuch as transport through an aquifer occurs at every type of site, it must be remembered that the exact results obtained for any site are highly dependent upon the actual hydrological and soil-water interaction parameters employed.

B4.5.5 Summary of Sensitivites

Any factor which can substantially delay contact of the radioactivity with the human population can be important in reducing the potential impact of liquid pathways. The primary hydrospheric factors which are important in discussing the impacts are any which effect the groundwater travel times and the soil-water interactions. Of secondary importance are the factors which relate to surface water transport. Generally of least importance are the factors which describe the sediment-water interactions.

References

[1] US Nuclear Regulatory Commission, "Liquid Pathway Generic Study", NUREG-0440, USNRC, Appendix B, 1978.

[2] M. Abramowitz and I.A. Stegun, *Handbook of Mathematical Functions with Formulas, Graphs and Mathematical Tables*, NBS, 1964.

[3] S.J. Niemczyk, "A Summary of Hydrological Information for Nuclear Power Plants," in preparation.

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*The impact of a given radionuclide is related to its concentration, its residence time, its radioactive decay rate and its biological effectiveness. Short flushing times tend to eliminate the effect of radioactive decay.

[4] E.V. Giusti and E. L. Meyer, "Water Consumption by Nuclear Powerplants and Some Hydrological Implications", Geological Survey Circular 745, USGS, 1977.

[5] H.C. Burkholder, M.O. Clonginer, D.A. Baker, and G. Jansen, "Incentives for Partitioning High-Level Waste", Nuclear Technology *31*, 202-217, 1976.

[6] US Nuclear Regulatory Commission, Reactor Safety Study (WASH-1400), NUREG-75/014, USNRC, Appendix VII, 1975.

[7] US Nuclear Regulatory Commission, "Liquid Pathway Generic Study", NUREG-0440, USNRC, Chapter 6, 1978.

[8] R.S. Booth, "A System Analysis Model for Calculating Radionuclide Transport Between Receiving Waters and Bottom Sediments," in *Environmental Toxicity of Aquatic Radionuclides: Models and Mechanisms*, edited by M.W. Miller and J.N. Stannard, Ann Arbor Science Publishers, Ann Arbor, Michigan, 1976.

[9] A. Lerman and G.J. Brunskill, "Migration of Major Constituents from Lake Sediments into Lake Water and Its Bearing on Lake Water Composition," Limnology and Oceanography *16*,880-890, 1971.

[10] F.T. Manheim, "The Diffusion of Ions in Unconsolidated Sediments," Earth and Planetary Science Letters *9*, 307-309, 1970.

[11] R.H. Rainey, "Natural Displacement of Pollution from the Great Lakes," Science *155*, 1242, 1967.

[12] J.L. Hough, *Geology of the Great Lakes*, University of Illinois, Urbania, 1958.

[13] US Nuclear Regulatory Commission, "Methods of Estimating Aquatic Dispersion of Effluents from Accidental and Routine Reactor Releases for the Purpose of Implementing Appendix I,"Regulatory Guide 1.113, Revision 1, USNRC, 1977.

[14] D.W. Pritchard, "What is an estuary: physical viewpoint", in Estuaries, Publ. No. 83 Amer. Assoc. Advan. Sci., 3, 1967.

[15] National Academy of Sciences, *Radioactivity in the Marine Environment*, National Academy of Sciences, Washington, 1971.

[16] M. M. Nichols, "Sediments of the James River Estuary," Geol. Soc. Amer. Mem. *133*, 169-212, 1972.

[17] R. H. Meade, "Transportation and Deposition of Sediments in Estuaries," Geol. Soc. Amer. Mem. *133*, 91-120, 1972.

[18] B. W. Nelson, "Biogeochemical Variables in Bottom Sediments of the Rappahannook River Estuary," Geol. Soc. Amer. Mem. *133*, 417-451, 1972.

[19] B.H. Ketchum and D.J. Keen, "The accumulation of river water over the continental shelf between Cape Cod and Chesapeake Bay," from Papers in Marine Biology and Oceanography, Suppl. to Vol. 3 of Deep Sea Res., 346, 1955.

[20] H. Stommel and A. Leetma, "Circulation on the Continental Shelf," Proc. NAS *69*, 3380, 1972.

[21] E.F. Gloyna (Chairperson), "Hydrologic Transport of Radionuclides," in *Proceedings of a Workshop on the Evaluation of Models Used for the Environmental Assessment of Radionuclide Releases*, Gatlinburg, TN, September 6-9, 1977 (CONF-770901). 33-72, 1977.

[22] J.E. Campbell, R.T. Dillon, M.S. Tierney, H.T. Davis, P.E. McGrath, F.J. Pearson, H.R. Shaw, J.C. Helton, and F.A. Donath, *Risk Methodology for Geologic Disposal of Radioactive Waste: Interim Report*, NUREG/CR 0458, SAND78-0029, Sandia Laboratories, 192-226, October, 1978.

[23] R.L Iman, J.C. Helton, and J.E. Campbell, *Risk Methodology for Geologic Disposal of Radioactive Water: Sensitivity Analysis Techniques*, NUREG/CR-0390, SAND78-0912, Sandia Laboratories, Appendix A, October 1978.

[24] C.S. Shih and E.F. Gloyna, *Radioactivity Transport in Water - Mathematical Model for the Transport of Radionuclides*, University of Texas, 1967.

[25] Y. Onishi, "Finite Element Models for Sediment and Contaminant Transport in Surface Waters. Transport of Sediment and Radionuclides in the Clinch River," BNWL-2227, Battelle Pacific Northwest Laboratories, July, 1977.

[26] Y. Onishi, "Mathematical Simulation of Sediment and Radionuclide Transport in the Columbia River," BNWL-2228, Battelle Pacific Northwest Laboratories, August, 1977.

[27] Y. Onishi, E.M. Arnold, R.J. Serne, C.E. Cowan, F.L. Thompson, and D.W. Mayer, "Mathematical Simulation of Sediment and Contaminant Transport in Surface Waters," NUREG/CR-0658, PNL-2902, Pacific Northwest Laboratories, January 1979.

[28] R.D. Richtmeyer and K.D. Morton, *Difference Methods for Initial-Value Problems*, 2nd Ed., Interscience Publishers, New York, NY, 1967.

[29] V.T. Chow, (Ed.), Handbook of Applied Hydrology, McGraw-Hill, New York, 1964.

[30] L.B. Leopold, M.G. Wolman, and J.P. Miller, *Fluvial Processes in Geomorphology*, W.H. Freeman, San Francisco, CA, 1964.

[31] D.K. Todd, (Ed.), *Water Encyclopedia*, Water Information Center, Inc, Huntington, NY, 1970.

[32] R.J. Morton, D.J. Nelson, and F.L. Parker, *Comprehensive Report of the Clinch River Study*, ORNL-4035, Oak Ridge National Laboratory, April 1967.

[33] W.B. Borland, "Reservoir Sedimentation," in *River Mechanics*, Vol. II, H.W. Shen (Ed.), Water Resources Publications, Fort Collins, Colorado, 1971.

[34] M.A. Churchill, J.S. Cragwall, R.W. Andrew and S.L. Jones, *Concentrations, Total Stream Loads, and Mass Transport of Radionuclides in the Clinch and Tennessee Rivers*, ORNL-3721 Suppl 1, Oak Ridge National Laboratory, August 1965.

[35] P.H. Carrigan, R.J. Pickering, T. Tamura and R. Forbes, *Radioactive Materials in Bottom Sediments of Clinch River: Part A, Investigations of Radionuclides in Upper Portion of Sediment*, ORNL 3721, (Suppl. 2A), Oak Ridge National Laboratory, March 1967.

[36] R.J. Morton, (Ed.), *Status Report No. 5 on Clinch River Study*, ORNL-3721, Oak Ridge National Laboratory, October 1965.

[37] R.D. Shull and E.F. Gloyna, *Radioactivity Transport in Water-Simulation of Sustained Releases to Selected River Environments*, CRWR 25, University of Texas at Austin (undated).

[38] H.B. Fischer, "The Mechanics of Dispersion in Natural Streams", J. Hydr. Div., Proc. ASCE, *93*, (HY6): 187-216, 1967.

[39] D.N. Edgington, M.A. Wahlgren, and J.S. Marshall, "The Behavior of Plutonium Aquatic Ecosystems: A Summary of Studies on the Great Lakes," from *Environment Toxicity of Aquatic Radionuclides: Models and Mechanisms*, edited by M.W. Miller and J.N. Stannard, Ann Arbor Science Publishers, Ann Arbor, Michigan, 1976.

[40] J.A. Robbins and D.N. Edgington, "Determination of Recent Sedimentation Rates in Lake Michigan Using Pb-210 and Cs-137," Geochimica and Cosmochimica Acta 39, 285-304, 1975.

ADDENDUM B1

Factors Affecting Distribution Coefficients

Chemical species in solution in groundwater can undergo many different reactions, both with other species in solution and with the soil through which they are passing. Among the possible interactions are physical adsorption, ion exchange, precipitation, colloid formation, complex formation and fixation. For radionuclide transport, the observable effect of all of these physicochemical processes is to delay the movement of the nuclide relative to the movement of the groundwater. The parameter which describes this retardation is the distribution coefficient, K_d. This addendum describes the factors affecting the values of this parameter.

Adsorption and ion exchange both result from reaction of a species in solution with a soil particle. After the process is reversed, the species once again moves along with the solution. The process of adsorption (or exchange) and its reversal are repeated throughout the motion of the species. Inasmuch as the soil particle is essentially fixed in space, the net effect is to slow down the reacting species relative to the water.

Precipitation results in the removal of a species from solution due to solubility considerations. Eventually the precipitated material can be redissolved. However, inasmuch as low solubilities were responsible for the initial precipitation, the redissolution is expected to proceed slowly. Consequently, the net flow of those species which tend to precipitate is relatively slow.

Colloid formation is another process which can greatly impede the flow of certain nuclides. Unless groundwater flow is through a comparatively open medium, the flow of colloidal species is essentially negligible. As for precipitates, the redissolution of colloids is generally relatively slow. As a result, colloid formers, such as the lanthanides, tend to have comparatively long travel times.

Complexation can be very important inasmuch as it can dramatrically increase the amount of a given element which can be in solution. It can increase the amount over that expected from simple solubility considerations by up to several orders of magnitude. Furthermore, complexation allows great variation in the possible charge states in which the species can exist. These charge states are critical in determining the amount of ion exchange which occurs. If neutral complexes are formed, then the species will not be ion-exchanged and may be passed through the soil with the groundwater. Similarly, if anionic complexes are formed for any species, then the amount of ion exchange will probably be greatly reduced.

Important chemical factors determining the interactions undergone, and hence the distribution coefficients, include the groundwater's ionic strength, chemical composition, pH and reduction (or oxidation) potential. These same chemical factors for the soil are also relevant.

If any ion is present in non-trace amounts, it can be expected that its distribution coefficient will be lowered due both to a possible shortage of exchange sites and to non-equilibrium adsorption conditions. Any other species present in solution in non-trace amounts can be important for ion exchange if it competes with a given ion for exchange sites. In general, chemically similar species such as Na^+ and K^+ can be expected to vie for the same sites. Therefore, the distribution coefficient of any species can be greatly lowered by non-trace amounts of similar species.

The pH is a measure of the H^+ content of the solution; that is, it is essentially a measure of the acidity of the solution. The reduction potential is a measure of the tendency to gain or to lose an electron. The pH and the reduction potential determine the thermodynamically preferred chemical state of a species. For example, depending on the conditions, Se can exist in solution at least as H_2SeO_3, $(HSeO_3)^-$, or $(SeO^3)^{-2}$. The tendency for a radionuclide for interaction is very dependent upon its chemical state. For example, if conditions favored the formation of H_2SeO_3, there would be no ion exchange of selenium with the soil. For certain soils, the ion exchange mechanism is such that only $(HSeO_3)^-$ is adsorbed;[1] thus there would probably also be insignificant exchange if $(SeO_3)^{-2}$ were the favored species.

As was indicated, the thermodynamic considerations determine which oxidation state (e.g., H_2SeO_3, $(HSeO_3)^-$ or $(SeO_3)^{-2}$) is preferred. However, the rate at which the preferred species is produced may be very slow. Consequently, the thermodynamically favored state may not be the dominant one in solution. The kinetics of any interactions must be considered along with the thermodynamics.

The preferred chemical forms are often quite different in reducing and oxidizing conditions. In well-mixed surface water and in groundwater near the ground surface, oxidizing conditions are expected. However, in stagnant water, in lower sediment layers and in regions well below the ground surface, reducing conditions are often present. Passage from a region of a given reduction potential to a region of a somewhat different potential can dramatically alter the preferred form of certain elements. As a result, the preferred chemical form of an element can vary as a function of location. For example, an element which was in solution in one region might rapidly precipitate out of solution as it attempted to move into a region with a different reduction potential; as a result, a large deposition of the material might be found at the reduction potential interface. If the kinetics of the change were slow, a more gradual precipitation could occur; consequently, rather than a sharply defined deposition, a spatially somewhat diffuse accumulation would result.

If several forms of a species are possible which differ at least in part by the H^+ or OH^- content, then the following generalization can be made: if a species tends to be present only as a cation (positive ion), then its distribution coefficient will usually be increased by a decrease in pH; in contrast, if a species tends to be present as an anion (negative ion), then its distribution coefficient will usually decrease with a decrease in pH.

In addition to the chemical factors mentioned previously, the physical state of the dissolved species is also important in determining the distribution coefficient. In general, those ions which are held most tightly after being taken out of solution by ion exchange are those which have both the smallest hydrated radii (largest unhydrated radii) and the highest charge; such species are most tightly bound electrostatically to exchange sites. For example, H^+ is among the ions which is least retained by the soil.

The character of the soil is also important in determining distribution coefficients. Certain soils have a much higher ion exchange capacity than others. For example, those with high clay and/or organic contents tend to have relatively large exchange capacities. Therefore, they tend to delay the passage of certain ions much more than some other soils. Furthermore, certain clays tend to irreversibly trap (fix) some species such as Cs^+.

Certain hydrogeological factors can also be relevant in determining the distribution coefficients. In particular, the rate of groundwater flow and any hydrogeological heterogeneities can be important. If the rate of groundwater flow is very high, there can be insufficient time for equilibrium between the soil and the water to be reached; thus, the ions will tend to be passed through the soil more quickly than would otherwise be expected. It should be noted that what may be inadequate time for one species to establish equilibrium may be more than enough time for another species to reach steady-state behavior.[2]

The distribution coefficients are a measure of the results of several chemical and physical processes over a range of hydrogeological conditions. If obtained in the laboratory, such coefficients do not take into consideration that groundwater transport occurs through a nonisotropic heterogeneous medium. If obtained in the field, such coefficients apply to only the specific location and set of experimental conditions for which they were determined. Distribution coefficients measured at a given site can differ by orders of magnitude from those appropriate for any other site. Therefore, it is important to employ site-specific distribution coefficients to adequately describe radionuclide movements at a given location.

References

[1] R. R. Frost and R. A. Griffin, "Effect of pH on Adsorption of Arsenic and Selenium from Landfill Leachate by Clay Minerals," Soil Science Society of American Journal *41*, 53-57, 1977.

[2] W. Back and B. B. Hanshaw, "Rates of Physical and Chemical Processes in a Carbonate Aquifer," in *Nonequilibrium Systems in Natural Water Chemistry*, Advances in Chemistry *106*, American Chemical Society, Washington, DC, 1971.

ADDENDUM B2

Details of Data Analysis for River System Model

B2.1 Sedimentation

Sedimentation in reservoirs was the subject of considerable study. Figure B2.1 shows a "typical" reservoir sediment profile.[33] The total volume of sediments in each of the Tennessee River reservoirs is known (Table B5). However, the irregularity of the larger reservoirs makes a study of sediment volume as a function of distance along the reservoir rather confusing. For example, sediment deposition can occur irregularly at many side inlets.

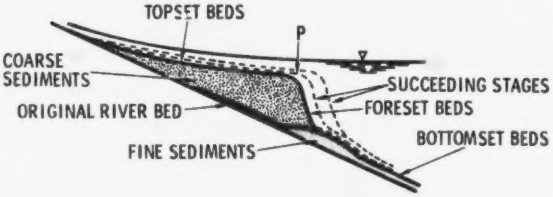
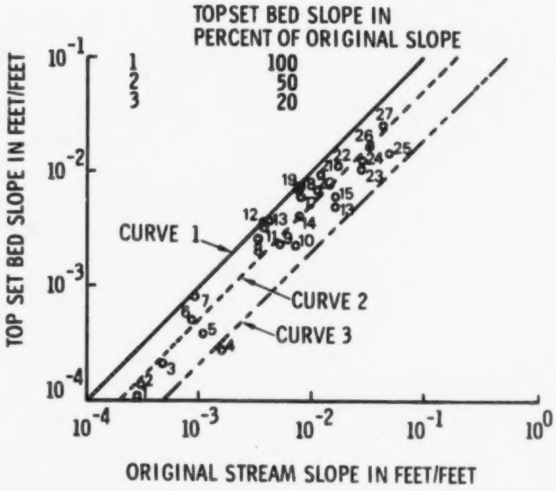


Figure B2.1 Profile of a Typical Reservoir Delta[33]

In general, the sediment beds in reservoirs can be divided into three types: topset, foreset and bottomset. Examples of these types are indicated in Figure B2.1. The ratio of topset slope to original stream slope is shown in Figure B2.2[33] for some "typical" reservoirs. This ratio varies from 0.5 to 1 for the depicted cases.

In this study, it was assumed that the topset bed contains 75 percent of the total sediment volume. The location of the foreset (P, Figure B2.1) was calculated for a number of the longer reservoirs under the assumption that the topset slope was half the original slope (using the same average slope for the entire backwater length). This leads to sedimentation distributions typified by Figure B2.3a. An assumption that the topset slope is nearly equal to the original slope results in distributions typified by Figure B2.3b.

Even though a topset delta probably builds up in the upper section of the reservoir, this normally receives the coarser sediments; the section closer to the dam (the bottomset bed) receives the finer sediments, as shown in Figure B2.1. It is known[34] that radioactivity tends to be more strongly associated with the finer sediments. Therefore, although the



Legend (Figure B2.2)
Location of Plotted Points

Reservoir	Stream
1. Dennison	Washita & Red Rivers
2. Dennison	Washita & Red Rivers
3. McMillan	Pecos River
4. Avalon	Pecos River
5. Guernsey	North Platte River
6. Altus (67 survey)	North Fork Red River
7. Altus (48 survey)	North Fork Red River
8. Boysen	Fivemile Creek (Wyo.)
9. Verdi Div. Dam	Truckee River
10. Saeltzer	Clear Creek (Calif.)
11. Jemez Canyon (65 survey)	Jemez River
12. Boysen	Muddy Creek (Wyo.)
13. Jemez Canyon (59 survey)	Jemez River
14. Jones Creek	Missouri River Loess Hills
15. Keswick (59 survey)	Spring Creek (Calif.)
16. Keswick (57 survey)	Spring Creek (Calif.)
17. Tongue River	Tongue River
18. Elephant Butte	Rio Grande
19. Saeltzer	Clear Creek (Calif.)
20. Alfred Lage	Missouri River Loess Hills
21. C. A. Stiles	Missouri River Loess Hills
22. Sheep Creek	Sheep Creek
23. Thomas Hodkin	Missouri River Loess Hills
24. Fred Brown	Missouri River Loess Hills
25. William Esbeck	Missouri River Loess Hills
26. Fred Hollrah	Missouri River Loess Hills
27. Emma La Frontz	Missouri River Loess Hills

Figure B2.2 Relationship Between Topset Slope and Original Stream Bed Slope[33]

inlet delta may have a higher sedimentation rate, the radioactivity per unit volume of sediments is probably lower there. This might tend to make the net deposition of radioactive material more uniform throughout the length. As a result, the second distribution in Figure B2.3 is probably adequate, and so a single average sedimentation rate was used throughout each reservoir.

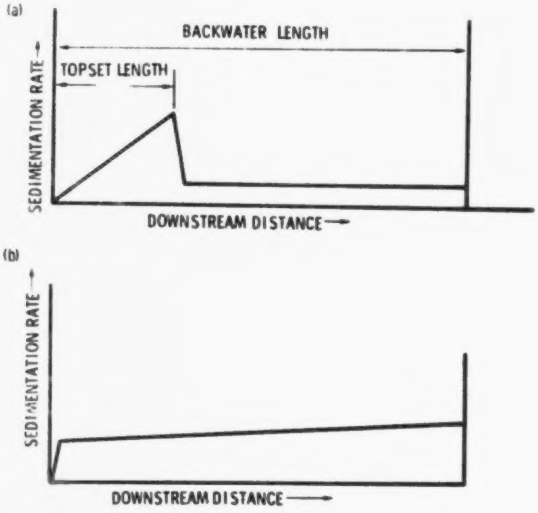


Figure B2.3 Sedimentation Distributions with (a) Topset Slope Less Than Original Slope and (b) Topset Slope Nearly Equal to Original Slope

Sedimentation rates for recent times were calculated from the data in Table B5, except that the rate for Hale's Bar Lake was modified. Each reservoir is assumed to have the appropriate long-term averaged sedimentation rate uniformly distributed through its length. The resultant estimates for the rates are presented in Table B5.

Hale's Bar Lake not only shows a net scour over a long period, but also shows the presence of radioactive sediments. The discrepancy could be due either to a very deep scour initially, followed by a slow buildup, resulting in a net scour, or to deep scour at the outfall from Chickamauga Dam and buildup only near Hale's Bar Dam. A low positive rate of 2 mm/year was assumed for Hale's Bar Lake.

An extensive survey has been conducted of radioactivity in the Clinch and the Tennessee Rivers.[35] Most of the measured radioactivity in the Tennessee has come from relatively steady releases into the Clinch over a rather extended period.[36] Inasmuch as the times required for water to pass through all the Tennessee reservoirs are relatively short, the entire set of these reservoirs can be considered to be essentially at equilibrium with respect to those releases. Because the releases have been fairly steady, the ratio of activity deposited in each of the lower reservoirs to that deposited at the mouth of the Clinch should remain fairly constant. Consequently, this system provides a good test case for a model.

Figure B2.4a shows the total activity ratios for Cs-137 in the Tennessee River reservoir sediments as measured in several different years.[35] The wide spread in the ratios can be attributed to two causes: spread in the accuracy of measurement, as indicated in Figure B2.5, and/or annual variations in both the flow and the sedimentation rates.[35] The corresponding calculated results for a steady check of Cs-137 (K_d = 27,000) into the Clinch River are shown in Figure B2.4b. The comparison with measured data is believed to be adequate.

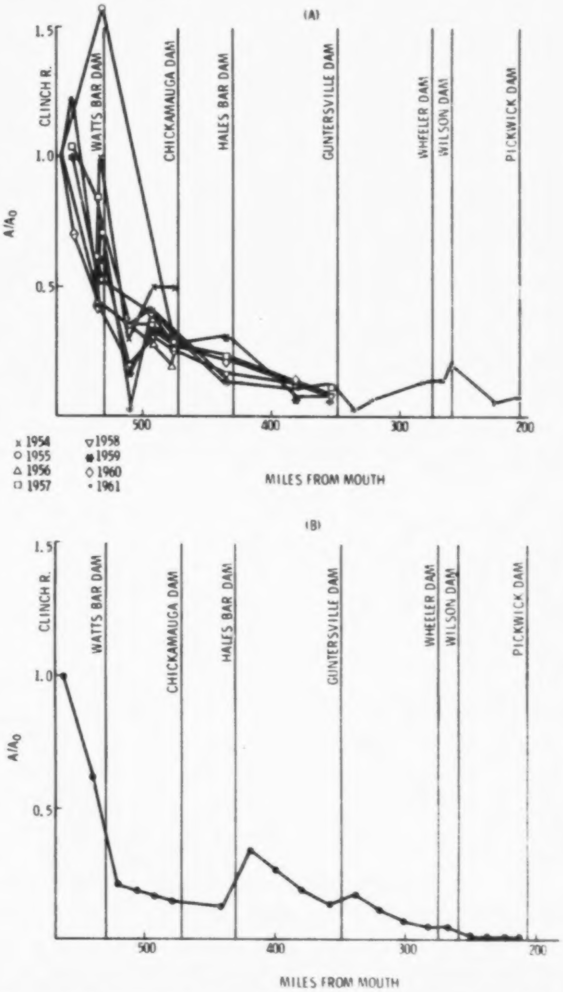


Figure B2.4 Cs-137: Ratio of Total Sediment Activity at the Indicated Location to the Total Sediment Activity near the Mouth of the Clinch River, with (a) Depicting Measured Values[35] and (b) Depicting Calculated Values.

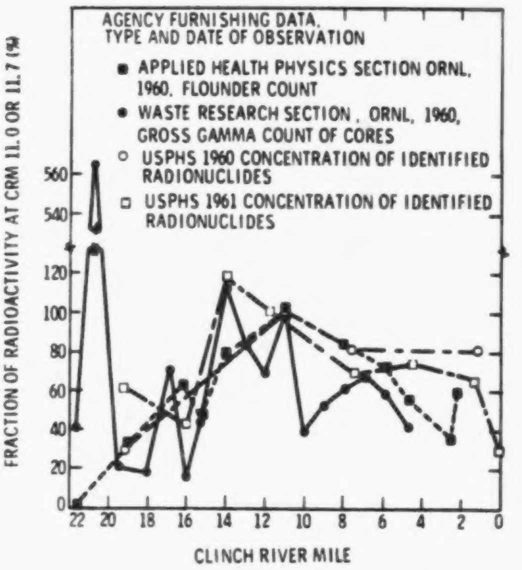


Figure B2.5 Comparison of Methods for Determining Longitudinal Distribution of Radioactivity[35]

Analogous measurements of Sr-90 in the reservoir sediments are shown in Figure B2.6a.[35] The model predictions for Sr-90 ($K_d = 2700$) are shown in Figure B2.6b. Considering the poor quality of the data, there is certainly no cause to refine the model for Sr-90.

Sedimentation in free-flowing river sections is of two types, floodplain aggradation and bed aggradation. The upper Missouri has shown a net bed degradation over the last few years.[30] Neither the Missouri or the Mississippi have been studied long enough to determine whether a long-term stable aggradation is in process. However, the lengthening of the Mississippi delta and the need for frequent dredging in the Mississippi and lower Missouri would seem to indicate some buildup.

For want of a better value, a sedimentation rate of 0.1 mm/yr has been used in all free-flowing river sections. This value is so low that sediment contamination in rivers is dominated by diffusive transfer.

B2.2 Suspended Sediment Load

The suspended sediment loads of rivers vary greatly, both from river to river and from one part of a river to another part. Rivers running through the farming regions, especially those having a high flow velocity, normally pick up heavy concentrations of sediment relative to rivers running through forested regions. The coarser fraction of sediment is usually deposited in the upper reaches, so that the percentage of fines tends to become higher in the lower reaches.

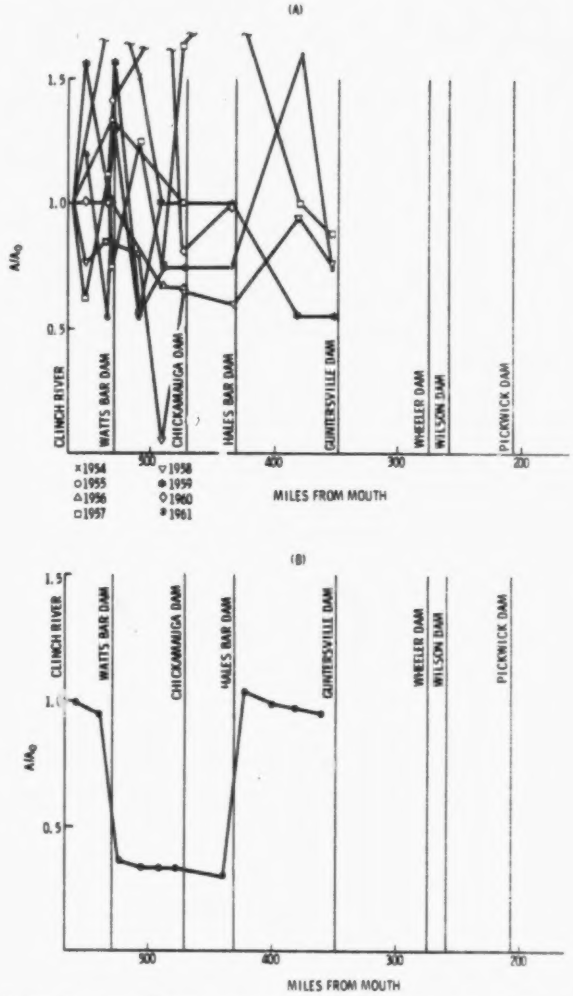


Figure B2.6 Sr-90: Ratio of Total Sediment Activity at the Indicated Location to the Total Sediment Activity near the Mouth of the Clinch River, with (a) Depicting Measured Values[36] and (b) Depicting Calculated Values.

The Missouri River delivers approximately 2.18×10^8 metric tons (MT) of sediment to the Mississippi anually, and the Mississippi, near its mouth, carries about 2.58×10^8 MT anually.[31] The sediment load of the lower reaches of the Tennessee is not known; the concentration in the Clinch River embayment of Watts Bar Lake averages 55 mg/1,[32] a weight concentration of 5.5×10^{-5}. If the sediments have an average specific gravity of 2.2, the volumetric concentration is about 2.5×10^{-5} at this point. Sedimentation in reservoirs undoubtedly reduces the concentration downstream. A similar southeastern river, the Flint River of Georgia, has an approximate volumetric concentration of 5.8×10^{-5}. By contrast, the Mississippi River has an average volumetric concentration of

2.3 × 10^{-4} near its mouth and the Missouri River has an average volumetric concentration of 1.6 × 10^{-3} near its mouth. The relative concentrations of sediment decreases downstream in all these rivers.

B2.3 Velocities and Dimensions

The width, the depth, and the magnitude of the velocity of streams and rivers are related to flow rate variations by[30]

$$W \propto (\dot{q})^{b},$$
$$D \propto (\dot{q})^{f},$$
$$V \propto (\dot{q})^{m}.$$

The exponents b, f, and m vary both from river to river and within a given river.

Figure B2.7[30] shows width, depth, and average velocity for a number of rivers as a function of volumetric flow. Case 7 in that figure was used to obtain equations for the width, the depth, and the average velocity for the Mississippi:

$$W = 1.286 \times 10^{-3} (\dot{q})^{0.5},$$
$$D = 1.847 \times 10^{-4} (\dot{q})^{0.41},$$
$$V = 4.21 \times 10^{6} (\dot{q})^{0.09},$$

where W = width in meters, D = depth in meters, and V = velocity in m/yr. An average of cases 4 and 5 was used for the Missouri. Both of the rivers represented in those cases flow through the High Plains like the upper Missouri. Although the lower Missouri flows through the Low Plains region, as does the Mississippi, the same relationships were used for the entire Missouri River:

$$W = 2.74 \times 10^{-3} (\dot{q})^{0.481},$$
$$D = 6.084 \times 10^{-5} (\dot{q})^{0.45},$$
$$V = 5.998 \times 10^{6} (\dot{q})^{0.069}.$$

This is reasonable inasmuch as the average values of the exponents for width, depth, and velocity for all Midwest Rivers are approximately 0.5, 0.4, and 0.1, respectively.[30]

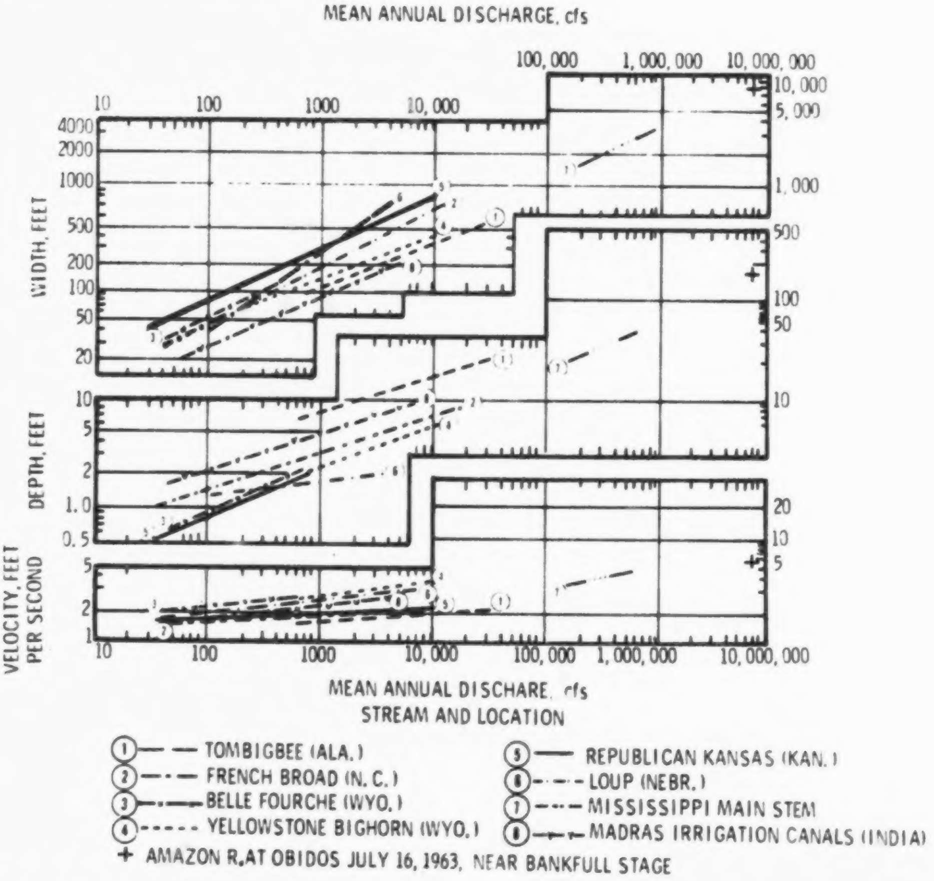


Figure B2.7 Width, Depth and Velocity in Relation to Mean Annual Discharge as Discharge Increased Downstream in Various River Systems[30]

Similar data were not found for the Tennessee River. However, almost the entire length of the Tennessee River is dammed, and is therefore modeled by reservoirs. For the very short remaining lengths, the Mississippi formulae are used. Because of the short length involved, it is expected that any error due to this procedure will be negligible.

In addition to the changes that occur in width, depth, and velocity as mean flow increases in the downstream direction, other changes in these parameters occur at any given location due to seasonal flow variations. For these changes, the relationships can be written as

$$\frac{W}{\overline{W}} = \left(\frac{\dot{q}}{\bar{q}}\right)^{b'}, \quad \frac{D}{\overline{D}} = \left(\frac{\dot{q}}{\bar{q}}\right)^{f'},$$

$$\frac{V}{\overline{V}} = \left(\frac{\dot{q}}{\bar{q}}\right)^{m'},$$

where $\overline{W}$, $\overline{D}$, and $\overline{V}$ are the width, the depth and the magnitude of the velocity at seasonally-averaged flow, $\bar{q}$.

All three exponents change when the flow changes from "bankfull" to over bankfull. Bankfull flow has a recurrence interval of about 1-5 years, and is about 7 times the mean annual discharge for rivers in the Eastern US and about 2.5 times the mean annual discharge for rivers in the High Plains.[30]

The specific values for the river systems under consideration could not be found. The average for less than "bankfull" conditions at 158 gaging stations in the US are:[30]

$b' \sim .12$,
$f' \sim .45$,
$m' \sim .43$.

From this it can be seen that for less than bankfull flow, the width generally increases relatively little during periods of high flow. This is not true for certain rivers in the Western US with cohesionless banks. For such rivers, the exponents are approximately

$b' \sim .5$,
$f' \sim .27$,
$m' \sim .23$.

Both the Tennessee and the Mississippi Rivers are of the former type; that is, the width increases relatively little with increase in flow. This may also be true of the Lower Missouri; the Upper Missouri is probably of the latter type, however.

In the interest of the uniformity, calculations have been made only with models of the former type, that is, for b′ very small. The values used are

$b' = 0$,
$f' = m' = 0.5$.

The exponent b′ has been set to zero for simplicity; a value of 0.12 would give only a 12% increase in width for a 150% increase in flow.

All the models just discussed for dealing with the changes of parameters in the downstream direction and the changes at a given location are similar to those used by Shull and Gloyna.[37]

B2.4 Seasonal Flow Variations

Flooding and low-water periods in the river systems were handled by a simplified procedure. Figure B2.8 shows the seasonal variation of runoff in some typical US rivers.[31] Peaks and valleys of the runoff occur at different times in different regions. However, in almost all regions of interest, the peak runoff is in spring and the minimum is in late summer. It can be seen that the maximum is more sharply peaked than the minimum.

Maximum discharge of the lower Mississippi is as much as 3.71 times the average.[37] Two-thirds of this (2.48)[*] was taken as the ratio of the peak to the average flow for all systems. Low discharge at the same point in the lower Mississippi has been as little as 0.178 times average. Twice this extreme was taken as the annual minimum flow.

The spring flood was assumed to rise to a peak and fall back to normal over a two-month interval. The low-water cycle was assumed to occur over a 7.6 month period. The entire cycle, as shown in Figure B2.9, was applied to all rivers in the system simultaneously. It should be pointed out that any seasonal flow pattern can be easily used in the model.

A time-dependent flow pattern, such as that shown in Figure B2.9 is given as input data. At each time, a "relative flow factor," $f_{rf} = \frac{\dot{q}(t)}{\bar{q}}$, is computed. $\bar{q}$ is the seasonally averaged flow. The velocities and depths are then computed according to

$V(t) = f_{rf}^{\,m'}\,\overline{V}$,
$D(t) = f_{rf}^{\,f'}\,\overline{D}$,

where $\overline{V}$ and $\overline{D}$ are the velocity and depth, respectively, at the seasonally averaged flow, and f′ and m′ are as given in the previous subsection.

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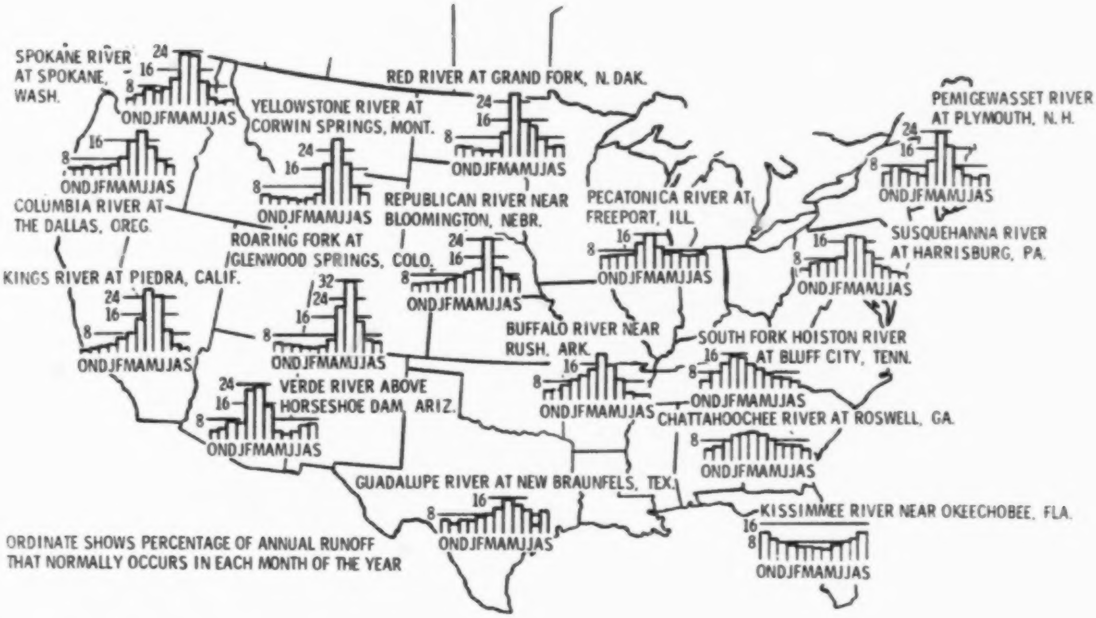
[*] This volumetric flow is approximately one-third of "bankful" flow for most rivers in the Eastern US, and approximately equal to "bankful" flow for rivers in the High Plains.[30]


SPOKANE RIVER AT SPOKANE, WASH.
YELLOWSTONE RIVER AT CORWIN SPRINGS, MONT.
RED RIVER AT GRAND FORK, N. DAK.
PEMIGEWASSET RIVER AT PLYMOUTH, N. H.
COLUMBIA RIVER AT THE DALLAS, OREG.
REPUBLICAN RIVER NEAR BLOOMINGTON, NEBR.
PECATONICA RIVER AT FREEPORT, ILL.
ROARING FORK AT GLENWOOD SPRINGS, COLO.
SUSQUEHANNA RIVER AT HARRISBURG, PA.
KINGS RIVER AT PIEDRA, CALIF.
BUFFALO RIVER NEAR RUSH, ARK.
SOUTH FORK HOISTON RIVER AT BLUFF CITY, TENN.
VERDE RIVER ABOVE HORSESHOE DAM, ARIZ.
CHATTAHOOCHEE RIVER AT ROSWELL, GA.
GUADALUPE RIVER AT NEW BRAUNFELS, TEX.
KISSIMMEE RIVER NEAR OKEECHOBEE, FLA.
ONDJFMAMJJAS
8
16
24
32

ORDINATE SHOWS PERCENTAGE OF ANNUAL RUNOFF THAT NORMALLY OCCURS IN EACH MONTH OF THE YEAR

Figure B2.8 Seasonal Variation of Runoff in the United States[31]

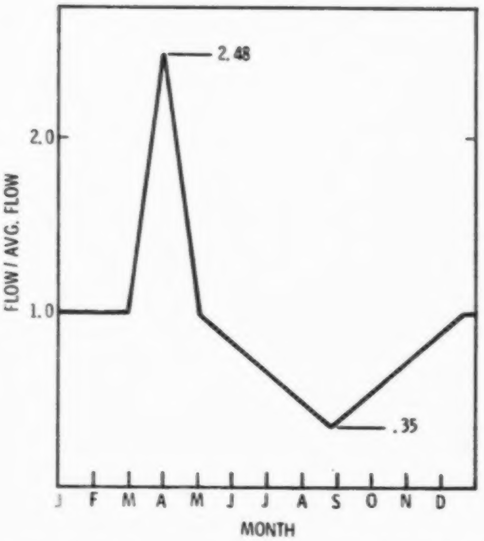


Figure B2.9 Seasonal Flow Variation Used in Calculations

B2.5 Dispersion Coefficients

The principal physical cause of dispersion in the free-flowing stretches of rivers is variation in velocity. For example, local instantaneous velocities as high as 20 mph have been measured in portions of the Mississippi, in which the average velocity is only about 5 mph; backwater sloughs can be found along the same portions in which the velocity is close to zero. Significant dispersion also results from smaller but more common variations (Figure B2.10).[33]

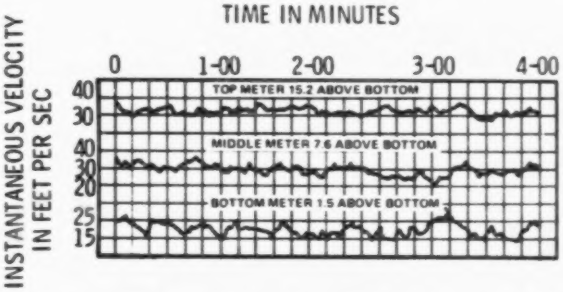


Figure B2.10 Velocity in the Mississippi River at Various Depths[33]

In view of the facts that (a) the one-dimensional dispersion coefficient can only give a crude approximation of dispersion, (b) dispersion has a very small influence on the results, and (c) the measurements required for refinement of the coefficient are not feasible, it is not possible or worthwhile to evaluate the coefficient very accurately. The values employed for the dispersion coefficient have been obtained by using the velocity-dependent simplified Fischer formula.[38]

ADDENDUM B3

Dispersion in a River System Model

If the radionuclide concentration is suddenly increased at the inlet to a reservoir, the concentration at the outlet will not immediately change. After an appreciable delay, the concentration at the outlet will slowly begin to rise. Eventually, if the inlet concentration remains constant, the outlet concentration will also approach a constant value. In the absence of any significant sinks or sources, the concentration at the outlet will approach the concentration at the inlet. The gradual rise of the concentration at the outlet is due to dispersion. The principal cause of this physical dispersion in an actual reservoir is the variability of the longitudinal velocity of the water as a function of location. Other effects, such as eddy mixing, also play a role.

If the reservoir is modeled as a single mixed tank and the inlet concentration suddenly rises, the outlet concentration immediately begins to rise, eventually approaching the inlet concentration. The outlet concentration for a single tank can be written as

$$c_o = c_i\,(1 - e^{-qt/V})\,,$$

where c_i is the inlet concentration, q is the flow rate, and V is the reservoir volume. The quantity V/q is the average residence time in the reservoir. The relatively rapid rise in the calculated outlet concentration, which can be called "model dispersion," is a mathematical effect completely unrelated to the physical dispersion. If the differential equation for the concentration is solved numerically, another dispersive effect—numerical dispersion—is added to the model dispersion. The numerical dispersion is normally insignificantly small in comparison with either physical or model dispersion.

If a single reservoir is modeled as a series of mixed tanks, with each tank receiving its input from the preceding tank and discharging to the following tank, the nature of the model dispersion changes. If there are n tanks, each of equal volume, the outlet concentration due to a sudden rise in the inlet concentration is

$$c_o = c_i\left\{e^{-n\dot{q}t/V}\left[\frac{\sum_{r=0}^{n-1}(-1)^{n-r}\left(\frac{-n\dot{q}t}{V}\right)^{n-1-r}}{(n-1-r)!}+1\right]\right\}.$$

The results are shown in Figure B3.1 for different numbers of tanks. As the number of tanks becomes very large, the flow becomes more "plug-like"; that is, the radioactivity does not appear at the outlet until after a delay time $V/\dot{q}$, at which time the concentration suddenly rises to the inlet concentration value. This behavior is not physically realistic. The outlet concentration should start to rise slowly at some time less than $V/\dot{q}$, and eventually become equal to the inlet concentration. The model behavior for a single tank is likewise unrealistic. It is impossible for the outlet concentration to begin to rise immediately in response to the inlet concentration.

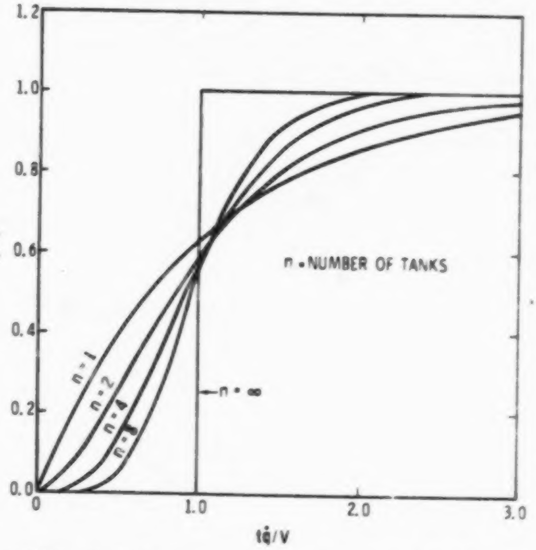


Figure B3.1 Outlet Concentration in Response to a Sudden Rise in the Inlet Concentration

Some middle ground between the single tank model and the plug-like model is required. In order to choose the optimum number of tanks to use, we would have to know the dispersive characteristics of each reservoir. These are physical details which it would be difficult and economically infeasible to measure.

APPENDIX C

[redacted]

APPENDIX C

C6 Summary of Results, Using the Foodweb Model

Author: S. J. Niemczyk

Appendix C has been issued as SAND80-1670, NUREG/CR-1597. Included in this report is only the summary for the appendix, along with some tables comparing the results of using the three aquatic food models considered in this study (see Appendix D).

For the three "generic" sites considered in this study,* the results** of the foodweb model, in terms of total population dose potentially received, are usually not substantially different from those obtained using the other two, less complicated aquatic food models (see Tables C1-C5). They depart most in situations with both short duration releases and relatively rapid flushing. In these situations, the departures between the predictions tend to be greatest when the organisms in the dominant foodchain(s) have net uptake and net elimination rates which are substantially different and/or the organisms have very slow uptake rates. Noticeable variations among the predictions are not seen in the large lake situation, except in the immediate vicinity of short-duration releases. Perceptible differences in the results are seen throughout both the estuary and coastal ecosystems for any short-duration releases. However, inasmuch as most of the releases to the hydrosphere after a meltdown accident are expected to be of relatively long duration, it can be said that for ecosystems such as the three considered here, the foodweb results do not usually depart significantly from those obtained using the other two models.

The three aquatic-food models are compared in Table C6. The foodweb model is necessary for an adequate description of any system in which a significant fraction of the radioactivity resides in any biological or detrital component of the ecosystem. It is also necessary to consider any ecosystem in which there are both relatively sudden changes of radioactivity in the water and slow transfers of activity in the dominant foodchain(s). The use of the foodweb model is further indicated whenever there are both significant concentration changes in the water and relatively slow and unequal net uptake and elimination rates. The foodweb model is also of potential importance whenever the exact time-dependent behavior of the radioactivity in the environment is relevant.

As the results presented in this appendix indicate, the concentration-factor model is not necessarily conservative. That is, there are situations, such as some of those just mentioned, in which the doses obtained by using that model are underestimates of the appropriate values. In such cases, the insights gained from using the foodweb model can be important.

It should be noted that the foodweb results at sites different than those considered here could depart more from the results of the other two models. Furthermore, the detailed information about the rate at which the dose would be delivered would always tend to differ much more from the predictions of the other two models than would just the estimated total doses at any site. However, the uncertainties in some of the more sensitive parameters, in particular, both in the uptake and elimination rate constants and in the detritus interaction parameters, would often handicap any decision as to whether any differences obtained were necessarily significant.*

---

*Only three sites (large lake, estuary, and ocean) were considered with the foodweb model.

**The results are presented in terms of concentration integrals. The values of these integrals are proportional to the population dose which would be received as a result of ingestion of contaminated aquatic food. (See Section B4 of Appendix B.)

*Inasmuch as the uptake and elimination rate constants are among the most sensitive parameters for either the foodweb model or any respectable time-dependent one-compartment aquatic-food model,[11] one of the most important indicated areas for further work on aquatic-food models is the assembly of all the relevant available information about such rates. Such a compilation would permit a much clearer overall picture of uptake and elimination to be developed. Therefore that information might result in a much improved description of the movement of radioactivity in any aquatic system. A similar data-gathering and analysis effort might be indicated for considering the role of detritus in the ecosystem. However, unlike the situation for uptake and elimination rates, much less direct information appears to be available about detrital interactions.

Table C1. Comparison of Concentration Integrals for the Three Aquatic Food Models After a Slow Release of Cs-137(a)

	Nearshore			Offshore		
Generic Site	Concentration-Factor Model	One-Compartment Time-Dependent Model	Foodweb Model	Concentration-Factor Model	One-Compartment Time-Dependent Model	Foodweb Model
Large Lake						
Finfish	1.5-11	1.5-11	8.5-12(b)	3.2-12	3.2-12	2.3-12(b)
Estuary						
Finfish	2.3-12	2.3-12	2.3-12	1.1-12	1.1-12	1.1-12
Crustaceans	1.9-12	1.9-12	2.0-12	9.4-13	9.4-13	9.7-13
Mollusks	1.9-12	1.9-12	2.0-12	9.4-13	9.4-13	9.7-13
Ocean						
Finfish	5.2-12	5.2-12	5.2-12	7.3-14	7.3-14	7.3-14
Crustaceans	4.4-12	4.4-12	4.5-12	6.1-14	6.1-14	6.0-14
Mollusks	4.4-12	4.4-12	4.5-12	6.1-14	6.1-14	6.0-14
Kelp	1.7-11	1.7-11	1.7-11	2.4-13	2.4-13	2.4-13

[a] A slow release is taken to be a uniform release of 1 Ci over a 10-year period; the concentration integrals are for 100 years after the start of the release.

[b] For slightly higher assimilation fractions for finfish uptake of either water or diet organisms, the foodweb results are essentially equal to those of the other models; the differences between the foodweb results and the others are due to the relatively high CF for Cs in freshwater finfish.

Table C2. Comparison of Concentration Integrals for the Three Aquatic-Food Models for an Instantaneous Release of Cs-137(a)

	Nearshore			Offshore		
Generic Site	Concentration-Factor Model	One-Compartment Time-Dependent Model	Foodweb Model	Concentration-Factor Model	One-Compartment Time-Dependent Model	Foodweb Model
Large Lake						
Finfish	1.5-11	1.5-11	5.6-12(b)	6.0-12	6.0-12	2.7-12(c)
Estuary						
Finfish	2.3-12	2.3-12	2.6-12(d)	1.1-12	1.1-12	1.1-12(d)
Crustaceans	1.9-12	1.9-12	7.0-12(d)	9.4-13	9.4-13	2.3-12(d)
Mollusks	1.9-12	1.9-12	7.0-12(d)	9.4-13	9.4-13	2.3-12(d)
Ocean						
Finfish	5.2-12	5.2-12	1.6-11(d)	7.3-14	7.3-14	7.3-14
Crustaceans	4.4-12	4.4-12	3.8-11(d)	6.1-14	6.1-14	6.0-14
Mollusks	4.4-12	4.4-12	3.8-11(d)	6.1-14	6.1-14	6.0-14
Kelp	1.7-11	1.7-11	1.7-11	2.4-13	2.4-13	2.4-13

[a] An instantaneous release is taken to be a uniform release of 1 Ci over a 0.1-year period; the concentration integrals are for 100 years after the release.

[b] For slightly higher assimilation fractions for finfish, the foodweb results exceed those of the other models by a factor of several; this result is due to relatively high CF for Cs in freshwater finfish.

[c] For slightly higher assimilation fractions for finfish, the foodweb results are identical to those of the other two models.

[d] For somewhat slower net uptake rates and/or slightly more rapid net elimination rates for this compartment than those used in obtaining these results, the foodweb model results are approximately equal to those of the other two models.

Table C3. Comparison of Concentration Integrals for the Three Aquatic-Food Models After a Slow Release of Ru-106(a)

	Nearshore			Offshore		
Generic Site	Concentration-Factor Model	One-Compartment Time-Dependent Model	Foodweb Model	Concentration-Factor Model	One-Compartment Time-Dependent Model	Foodweb Model
Large Lake						
Finfish	7.5-14	7.5-14	7.4-14	2.1-15	2.1-15	2.1-15
Estuary						
Finfish	1.9-13	1.9-13	1.9-13	8.5-14	8.5-14	8.5-14
Crustaceans	6.4-12	6.4-12	5.4-12	2.8-12	2.8-12	2.4-12
Mollusks	6.4-12	6.4-12	5.4-12	2.8-12	2.8-12	2.4-12
Ocean						
Finfish	5.2-13	5.2-12	5.2-13	4.3-15	4.3-15	4.3-15
Crustaceans	1.7-11	1.7-11	1.6-11	1.4-13	1.4-13	1.3-13
Mollusks	1.7-11	1.7-11	1.6-11	1.4-13	1.4-13	1.3-13
Kelp	1.7-10	1.7-10	1.7-10	1.4-12	1.4-12	1.4-12

(a) A slow release is taken to be a uniform release of 1 Ci over a 10-year period; the concentration integrals are for 100 years after the start of the release.

Table C4. Comparison of Concentration Integrals for the Three Aquatic-Food Models After an Instantaneous Release of Ru-106(a)

	Nearshore			Offshore		
Generic Site	Concentration-Factor Model	One-Compartment Time-Dependent Model	Foodweb Model	Concentration-Factor Model	One-Compartment Time-Dependent Model	Foodweb Model
Large Lake						
Finfish	7.5-14	7.5-14	2.9-13(b)	2.1-15	2.1-15	2.1-15
Estuary						
Finfish	1.9-13	1.9-13	2.2-13(b)	8.5-14	8.5-14	8.5-14
Crustaceans	6.4-12	6.4-12	9.7-12(b)	2.8-12	2.8-12	3.7-12(b)
Mollusks	6.4-12	6.4-12	9.7-12(b)	2.8-12	2.8-12	3.7-12(b)
Ocean						
Finfish	5.2-13	5.2-13	1.4-12(b)	4.3-15	4.3-15	4.3-15
Crustaceans	1.7-11	1.7-11	6.4-11(b)	1.4-13	1.4-13	1.3-13
Mollusks	1.7-11	1.7-11	6.4-11(b)	1.4-13	1.4-13	1.3-13
Kelp	1.7-10	1.7-10	1.7-10	1.4-12	1.4-12	1.4-12

(a) An instantaneous release is taken to be an essentially uniform release of 1 Ci over a 0.1-year period.

(b) This foodweb model result is approximately equal to the corresponding results of the other two models for somewhat slower net uptake rates and/or slightly faster elimination rates for this compartment than those used in obtaining this result.

Table C5. Comparison of Concentration Integrals for the Three Aquatic Food Models After a Release of Sr-90 into a Large Lake[a]

Finfish	Concentration-Factor Model	One-Compartment Time-Dependent Model	Foodweb Model
Nearshore - slow release[b]	2.3-13	2.3-13	2.2-13
Offshore - slow release[b]	8.8-14	8.8-14	8.6-14
Nearshore - rapid release[c]	2.3-13	2.3-13	1.0-12[d]
Offshore - rapid release[c]	8.6-14	8.6-14	8.7-14

[a] Sr-90 does not concentrate significantly in the flesh of most saltwater organisms; therefore, results are presented here for Sr-90 only for the single freshwater site considered.

[b] A slow release is taken to be a uniform release of 1 Ci over a 10-year period; the concentration integrals are for 100 years after the start of the release.

[c] A rapid release is taken to be an essentially uniform release of 1 Ci over a 0.1-year period; the concentration integrals are for 100 years after the start of the release.

[d] This foodweb model result is approximately equal to the corresponding results of the other two models for somewhat slower net uptake rates and/or slightly faster elimination rates for this compartment than those used in obtaining this result.

Table C6. Comparison of Aquatic-food Models

Feature	Model Concentration-Factor Model	One-Compartment Time-Dependent Model	Foodweb Model
Conservation of mass	No	No	Yes
Conservation of radioactivity	No	No	Yes
Time-dependent behavior:			
Water	Yes	Yes	Yes
Aquatic organisms	No	In one set of organisms only.	In all organisms.
Concentration factors	Always satisfied	Used to set limits.	Used to set limits.
Uptake and elimination rate constants	Not considered	u.r. = e.r.[a]	u.r. and e.r. independent, often result of complex set of interactions.
Data required	CF's only	CF's, plus e.r.'s	All parameters considered in Table C11 of SAND80-1670, NUREG/CR-1597.
Applicable situations	Approximate steady-state conditions in which the biota can cause only a small perturbation of the movement of the radioactivity in the environment (i.e., small enough biomasses, turnover rates, concentration factors, etc).	Any conditions in which the biota cause only a small perturbation of the movement of the activity and in which the uptake and/or elimination is slow only in at most one set of organisms.	All conditions.
Results	$CI_i = B_i \cdot CI_w$ and $C_i(t) = B_i \cdot C_w(t)$ for all t.[b]	If u.r. = e.r., then CI_i (this model) = CI_i (CF model) but the magnitudes and the temporal descriptions of the corresponding concentrations can be very different; if u.r. is actually greater than e.r., then the results of this model (both CI's and C's) can be too low; if u.r. is actually less than e.r., then the results can be too high.[c]	Both CI_i and $C_i(t)$ can depart substantially from those of the other two models whenever (1) the effective concentration in the water is changing relatively rapidly, (2) the biota significantly perturb the concentration in the water, or (3) the net uptake and/or elimination rates are comparatively slow.
Limitations	Of very restricted applicability.	Of rather restricted applicability.	Members of populations not adequately described; large number of parameters required.

[a] An improved one-compartment model without this restriction is presented in Reference 11.

[b] CI_i denotes the concentration integral for radionuclide i; C_i denotes the concentration for that radionuclide; t denotes the time.

[c] This assumes that the e.r. data are used to set both values.

6

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References

[1] G. E. Runkle, "A Biological Survey of the Aquatic Foodweb at Lake Michigan, the Chesapeake Bay and the Southern California Ocean," in preparation.

[2] M. G. Marietta, J. R. Wayland, G. E. Runkle, D. Jackson, J. Baker, and A. Kwok, "Aquatic Foodweb Radionuclide Migration Model," NUREG-CR-1453/SAND80-1047, Sandia Laboratories, Albuquerque, NM 1980.

[3] L. B. Slobodkin, "How to be a Predator," Amer. Zool. *8*, 43-52 (1968).

[4] M. B. Schaefer, "Economic and Social Needs for Marine Resources," in J. F. Brahtz, Editor, *Ocean Engineering*, Wiley, New York, 6-37, 1968.

[5] National Academy of Science, *Radioactivity in the Marine Environment*, National Academy of Science, Washington, DC, 1971.

[6] G. G. Polikarpov, *Radioecology of Aquatic Organisms*, Reinhold, New York, 1966.

[7] V. M. Klechkovskii, G. G. Polikarpov, and R. M. Aleksakhin, Eds., *Radioecology*, John Wiley and Sons, New York, 1973.

8. I. L. Ophel and J. M. Judd, "Experimental Studies of Radiostrontium Accumulation by Freshwater Fish from Food and Water- *Radioecological Processes*, Pergamon Press, 1965.

[9] F. Morgan, J. Mar. Biol. Assoc. U. K. 44, 259 (1964).

[10] J. A. Colin Nicol, *The Biology of Marine Animals*, Pitman and Sons Ltd., London, 1967.

[11] S. J. Niemczyk, "An Improved One-Compartment Aquatic Food Model for Considering Radionuclide Concentrations"in preparation.

APPENDIX D

Dosimetry and Health Effects

APPENDIX D

Dosimetry and Health Effects

Authors: S. J. Niemczyk
L. T. Ritchie (Shoreline Model)
W. B. Murfin (Addendum D2)

D1 Introduction

This appendix describes the estimation of the radiation dose received by the population after a core-melt accident. It also considers the approximation of the resultant health effects.

In Section D2, the models utilized for the calculations of the dose to an individual and the dose to the population are presented. In the discussion of individual dose, the dependence of the dose on time-dependent interactions in the environment is emphasized; in the discussion of population dose, the dependence of that dose on spatial location (distance from the accident site*) is also considered. Next, in Section D3, the health effects models are discussed. All the symbols used for the dosimetry models are listed in the first addendum presented at the end of this appendix. An improved water treatment model is discussed in the second addendum. Finally, some ingestion dose factors used in this report are presented in the third addendum. Related qualitative discussions of the factors which affect the movement of radioactivity along the various liquid exposure pathways and which therefore help to determine the resultant doses and health effects are given in Appendix E.

D2 Doses

D2.1 Individual Dose

Five types of exposure pathways are considered in this study: ingestion of drinking water, ingestion of aquatic foods, ingestion of irrigated foods, shoreline usage, and immersion in water. The population doses for all these liquid pathways ultimately depend on the radionuclide concentrations in the water. In an accident situation, these concentrations can be expected to vary substantially within any waterbody as functions of both time and location. In addition to the time-dependent behavior of the radioactivity in the water itself, three of the five most important pathways to the human population involve extensive time-dependent interaction of the nuclides with other parts of the environment before the exposure occurs. The three pathways are those involving aquatic foods, irrigated foods, and shoreline usage. For these pathways, the exposure is often not directly proportional to the concentration in the water. Instead, the concentration of radioactivity in any of these pathways is a derived function of some combination of the water concentration and an appropriate buildup (and/or removal) factor for the pathway.

In this study, the variations of the concentrations of radioactivity in the water as functions of time and space have been considered for all five pathways. Furthermore, for the three pathways involving extensive interactions with additional portions of the environment, time-dependent buildup in (and removal from) these pathways has been included. In addition, for the aquatic food pathway, a detailed description of the time dependence of the complete network of the major interactions with the environment has also been carried out. The time dependencies and the interaction networks of the other four pathways have been treated in much less detail.

With the exception of the formulae used for shoreline exposure and for aquatic food ingestion, the expressions utilized for estimating doses to an individual are essentially adapted versions of those presented in Regulatory Guide 1.109.[1] The expression for the shoreline exposure has been modified to include the effects of some weathering. The aquatic food exposure has been considered with three different models: a time-independent model, analogous to that presented in Regulatory Guide 1.109; a one-compartment time-dependent model, analogous to that utilized in NUREG-0440;[2] and, a multi-compartment time-dependent foodweb model, mentioned previously and discussed in detail in Appendix C and in a separate report, SAND80-1670 (NUREG/CR-1597). All the Regulatory Guide 1.109 expressions for doses have been extended to incorporate the variation of the radionuclide concentrations with time in a non-steady-state fashion.

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*Distance is taken to be as measured in terms of the relevant contaminated water body(ies).

D2.1.1 General Expression

The dose received by an individual between times 0 and t from exposure by means of pathway p can be written as

$$\text{dose}_p = U_p \sum_i d_{ip} f_{ip} e^{-\lambda_i t_p} \int_0^t C_{ip}(t')\, dt' , \qquad \text{(D2-1)}$$

where

U_p is the individual usage rate,
d_{ip} is the dose factor for isotope i and pathway p,
f_{ip} is the removal or modification factor for isotope i and pathway p,
λ_i is the radioactive decay constant for isotope i,
t_p is the delay time for pathway p, and
$C_{ip}(t)$ is the concentration of isotope i in pathway p.

The factor $(e^{-\lambda_i t_p})$ allows for decay during the period after removal from the waterbody and before exposure.

The concentration of radionuclide i in pathway p can often be approximated as

$$C_{ip}(t) \simeq \ldots e^{-\lambda_{ei} t} \int_0^t \ldots C_{iw}(t')\, e^{+\lambda_{ei} t'}\, dt',$$

where

$C_{iw}(t')$ is the concentration of radionuclide i in the water, and
λ_{ei} is an effective removal rate, that is, $\lambda_{ei} = \lambda_i + \lambda_w$, where λ_w is a physical removal or weathering rate.

Such an expression in a model for the pathway concentration permits gradual buildup in and removal from an exposure pathway.

D2.1.2 Specific Pathway Expressions

In the following subsections, the five pathways will be denoted by the following subscripts:

drinking water ingestion	d;
aquatic food ingestion	a;
irrigated food ingestion	i;
shoreline usage	s;
immersion or bathing	b.

D2.1.2.1 Drinking Water

The dose received by an individual from ingestion of drinking water between times 0 and t can be written as

$$\text{dose}_d = U_d \sum_i d_{ii} e^{-\lambda_i t_d} \int_0^t C_{id}(t')\, dt' , \qquad \text{(D2-2)}$$

where

U_d is the individual consumption rate for water,
d_{ii} is the ingestion dose factor for isotope i,
t_d is the time between removal of the water from the affected waterbody and consumption by the individual, and
$C_{id}(t)$ is the concentration of isotope i in the drinking water.

Most drinking water undergoes some type of processing before consumption. Reasonable delays caused by such processing vary from hours to days. This study assumes a uniform delay time of 1 day. (Because the processing delay is generally much less than the other delays encountered in movement through the hydrosphere, the dose received via drinking water can be expected to be rather insensitive to the value chosen for the delay time.)

Water treatment not only delays the water but it also results in the partial removal of certain radionuclides from the water. The amount of radioactivity removed depends on the type of treatment undergone.

Public water treatment facilities can be grouped into three types[3,4] on the basis of the purification processes which are employed:

1. Chlorination: results in no removal of radionuclides;
2. Sedimentation, filtration and chlorination: results in the removal of nearly all suspended radionuclides;
3. Coagulation, flocculation, sedimentation, filtration and chlorination: results in the removal of all suspended radionuclides and variable amounts of dissolved nuclides.

Consequently, to accurately assess the effects of a water purification system, one must know the amount of radionuclide present on suspended sediment, the amount of nuclide present in solution, and the type of water treatment facility performing the processing.

The amount of a radionuclide which passes through a given facility can be written as

$$A_i' = A_{iw}f_{iw} + A_{is}f_{is} \ , \qquad \text{(D2-3)}$$

where

A_{iw} = amount of isotope i initially in solution;
f_{iw} = fraction of i in solution which passes through the facility;
A_{is} = amount of i initially in suspension and;
f_{is} = fraction of i in suspension which passes through the facility.

Using this expression for the amount, the concentration in the processed drinking water can be written as

$$C_{id}(t) = \frac{A_{iw}f_{iw} + A_{is}f_{is}}{A_i} \cdot C_{iw}(t) \ , \qquad \text{(D2-4)}$$

where

A_i is the amount of isotope i initially in the water, and
$C_{iw}(t)$ is the concentration of isotope i in untreated water.

Some proposed values for the modification factors (fractions remaining) resulting from various levels of treatment are given in Tables D1 and D2.[5] Based on data available from the water processing facilities in Pasco and Richland, Washington, the fractions remaining in solution, and given in Table D2, appear to be most reasonable for a facility processing at level 3.

Table D1. Assumed Values for Fractions Remaining After Passage Through the Various Types of Water Processing Facilities

Facility Type	f_{iw}	f_{is}
1	1.0	1.0
2	1.0	0.0
3	see Table D2	0.0

Table D2. Drinking Water Retention Factors for Radioactivity in Solution

Element	Retention Factor
H	1.0
Co	0.2
Ni	0.2
Rb	0.9
Sr	0.2
Y	0.2
Zr	0.7
Nb	0.7
Mo	0.9
Tc	0.7
Ru	0.5
Rh	0.5
Pd	0.5
Ag	0.7
Cd	0.6
Sn	0.7
Sb	0.8
Te	0.8
I	0.8
Cs	0.9
Ba	0.4
La	0.2
Ce	0.2
Pr	0.2
Nd	0.2
Pm	0.2
Sm	0.2
Eu	0.2
U	0.7
Np	0.7
Pu	0.7
Am	0.7
Cm	0.7

The level of treatment employed depends upon the source of the water. For example, for water taken from an aquifer, one might not expect processing of drinking water beyond that of a Type 1 facility. However, for water taken from either a river or a small dammed reservoir with turbid water, one would expect processing at least at the level of a Type 2 facility due to the large amount of suspended sediment.

To be most conservative, one can assume that all the radioactive material resides in the solution phase and that all of it passes through the water treatment facility without removal. This study employs somewhat more realistic, yet still conservative, assumptions about both the state of the radioactive material and the effects of water treatment.

This study assumes that all the radioactive material withdrawn from aquifers and large lakes is in the solution phase but assumes that different levels of treatment are appropriate for these two types of waterbodies. Minimum treatment (Type 1) is assumed appropriate for water drawn from aquifers. At all large lake sites, 50 percent of the water is assumed to undergo either minimal or moderate treatment (Type 1 or Type 2) and the remaining 50 percent is taken to receive more extensive treatment (Type 3).

In contrast, for river and reservoir sites, not all of the radioactive material is assumed to be in solution. Rivers and reservoirs often have relatively large amounts of suspended sediment. Consequently, assuming that all the radioactive material were in solution could result in a significant over estimation of the amount of radioactivity which reached the human population through drinking water.

Rivers can be roughly divided into three categories according to their sediment loads as being either clear, average or turbid. The type of water treatment which is employed is taken to be a function of that sediment load. For this study, all drinking water from relatively clear rivers and reservoirs is assumed to undergo Type 1 treatment. Drinking water from river systems of average turbidity is taken to be treated by any of the three types of processing, with a population weighted average of the three being assumed for calculations (Table D3). All drinking water from turbid rivers and reservoirs is taken to undergo Type 3 treatment.

In addition to the activity removed from solution by processing, the total amount of activity in the river and reservoir water is taken to be lowered by removal of suspended solids and their adsorbed radionuclides. The sediment removal fractions assumed for river systems, and presented in Table D3, are probably conservatively low; Type 3 facilities actually remove essentially all suspended solids, and filtration alone (Type 2) removes as much as 95 percent. However, even the assumed fractions should greatly improve the realism of the consideration of drinking water taken from rivers. The details of the calculation of the reduction of radioactivity by sediment removal are presented in Addendum D2.

Table D3. Water Treatment Facilities for the 100 Largest U.S. Cities[6]

Treatment Type	Fraction of Population Served	Assumed Sediment Removal Fraction*
1	.27	0.0
2	.04	0.8
3	.69	0.9

*These values are used only for river systems.

The water treatments employed at all of the different types of sites are summarized in Table D4.* In particular, the fractions and the amounts of the initial activity remaining after treatment are given there.

D2.1.2.2 Aquatic Food

The dose received by an individual from ingestion of aquatic foods between times 0 and t can be written as

$$\text{dose}_a = \sum_m U_a^m \sum_i d_{ii} f_{im} e^{-\lambda_i t_m} \int_0^t C_{im}(t')\, dt' \ , \qquad \text{(D2-5)}$$

where

U_a^m is the individual consumption rate for aquatic food m,

f_{im} is the modification factor for isotope i and aquatic food m due to processing,

t_m is the delay time between removal of species m from the water and investion by the individual, and

$C_{im}(t)$ is the concentration of isotope i in aquatic food m.

The pathway involving aquatic foods has been treated at three levels in this study. The treatments differ in the way in which $C_{im}(t)$ is determined.

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*The effect of additional water treatment within individual homes is not included.

Table D4. Amounts of Radioactivity Sorbed Onto Sediments and in Solution Prior to Treatment and Fractions of These Amounts Remaining After Treatment

Site Type	A_{is}	F_{is}	A_{iw}	F_{iw}
aquifer	0	—*	A_i	1.0
large lake (any Great Lake)	0	—*	A_i	1/2 (value from Table D2) + 0.50
"clear" river (Tennessee River)	$\dfrac{A_i K_{di} 10^{-5}}{1 + (1 \times 10^{-5} \cdot K_{di})}$ **	0.90	$\dfrac{A_i}{1 + (1 \times 10^{-5} \cdot K_{di})}$	1.0
average river (Mississippi River)	$\dfrac{A_{iw} K_{di} \cdot 2.3 \times 10^{-4}}{1 + (2.3 \times 10^{-4} K_{di})}$	0.34	$\dfrac{A_i}{1 + (2.3 \times 10^{-4} \cdot K_{di})}$	0.69 (value from Table D2) + 0.31
turbid river (Missouri River)	$\dfrac{A_i K_{di} \cdot 1.6 \times 10^{-3}}{1 + (1.6 \times 10^{-3} \cdot K_{di})}$	0.10	$\dfrac{A_i}{1 + (1.6 \times 10^{-3} \cdot K_{di})}$	Value from Table D2

*At these sites, all the radioactivity is assumed to be in solution.

**The values used for the distribution coefficients, $\{K_{di}\}$, are given in Appendix B.

In the normally-used steady-state approach, the radioactivity present in the water and that present in organism m are assumed to be in equilibrium. Therefore the concentrations are taken to satisfy the relationship

$$C_{im}(t) = B_{im} C_{iw}(t), \tag{D2-6}$$

where

B_{im} is the concentration factor, or bioaccumulation factor, for isotope i in organism m.

This approach ignores the fact that it can take a significant period of time for the radioactivity in any organism to come to equilibrium with the radioactivity in the water. In an environment with a relatively rapidly changing concen tration of radioactivity in the water, equilibrium may never be reached.

In the second approximation, the single-compartment model,[2,7,8] the radioactivity in the organism approaches the concentration given above according to

$$C_{im}(t) = B_{im} \lambda_{bi} e^{-\lambda_{ei} t} (1 - e^{-\lambda_{bi} T})^{-1} \int_0^t C_{iw}(t') e^{\lambda_{ei} t'} dt', \tag{D2-7}$$

where

λ_{bi} is the biological (metabolic) decay rate,

λ_{ei} is the net elimination rate and is given by $\lambda_{ei} = \lambda_{bi} + \lambda i$, and

T is the average lifetime of species m.

This approach assumes that members of all the trophic levels not consumed by humans come to equilibrium instantaneously but that the species consumed by humans approach equilibrium relatively slowly, according to the above expression. Removal of radioactivity from the organisms is by radioactive decay, metabolic elimination and death. This approach ignores the fact that some aquatic food chains contain series of two or more species, each of which approach equilibrium behavior rather slowly; it also ignores the fact that aquatic organisms are part of a very complex interacting system called a foodweb.

In the third approximation, the foodweb model, the radioactivity approaches equilibrium as a function of time in every biological species considered. The entire foodweb network is included in the model. This approach is discussed in detail in Appendix C and in SAND80-1670 and is reviewed briefly in Chapter 5. (Comparisons of the three aquatic food models and of the results of using them are also presented in Appendix C and in SAND80-1670).

Realistic average values for the delay times for aquatic foods, t_m, are given in Table D5.[2] In this study, the fraction remaining after processing, f_{im}, is taken to be 0.31 for all vertebrate species, 0.45 for crustaceans, and 1.0 for all other invertebrate species and all aquatic plants; it is taken to be independent of the isotope. The derivation of these fractions is presented in Appendix E. The values used for the concentration factors, $\{B_{im}\}$, and for the biological (metabolic) decay rates, $\{\lambda_{bi}\}$, in the steady-state and in the one-compartment models are given in Tables D6[5] and D7,[9] respectively. It should be noted that the concentration factors for crustaceans, mollusks and finfish (given in Table D6) are for edible portions only.

Table D5. Aquatic Food Delay Times[2]

Type	Delay Time
Recreational Fish	1 day
Commercial Fish	
Fresh (8%)	C 3 days
Frozen (34%)	25 days
Canned (58%)	70 days

Table D6. Generic Bioaccumulation Factors (µCi/kg seafood per µCi/liter water)[5, (a)]

	Saltwater Factors				Freshwater Factors			
Element	Fish	Crus.	Moll.	Algae	Fish	Crus.	Moll.	Algae
H	1.0	1.0	1.0	1.0	.9	.9	.9	.9
Co	100.0	10000.0	300.0	100.0	50.0	200.0	200.0	200.0
Ni	500.0	100.0	100.0	100.0	100.0	100.0	100.0	50.0
Rb	30.0	50.0	10.0	10.0	2000.0	1000.0	1000.0	1000.0
Sr	1.0	1.0	1.0	20.0	30.0	100.0	100.0	500.0
Y	30.0	100.0	100.0	300.0	25.0	1000.0	1000.0	5000.0
Zr	30.0	100.0	100.0	1000.0	3.3	6.7	6.7	1000.0
Nb	100.0	200.0	200.0	100.0	30000.0	100.0	100.0	800.0
Mo	10.0	100.0	100.0	100.0	10.0	10.0	10.0	1000.0
Tc	10.0	100.0	100.0	1000.0	15.0	5.0	5.0	40.0
Ru	3.0	100.0	100.0	1000.0	10.0	300.0	300.0	2000.0
Rh	10.0	100.0	100.0	100.0	10.0	300.0	300.0	200.0
Pd	10.0	100.0	100.0	100.0	10.0	300.0	300.0	200.0
Ag	1000.0	5000.0	5000.0	1000.0	2.3	770.0	770.0	200.0
Cd	100.0	3000.0	10000.0	10000.0	200.0	2000.0	2000.0	1000.0
Sn	3.0	3.0	3.0	10.0	3000.0	1000.0	1000.0	100.0
Sb	1000.0	1000.0	1000.0	10000.0	1.0	10.0	10.0	1500.0
Te	10.0	10.0	100.0	1000.0	400.0	75.0	75.0	100.0
I	20.0	100.0	100.0	10000.0	15.0	5.0	5.0	40.0
Cs	30.0	50.0	10.0	10.0	2000.0	100.0	100.0	500.0
Ba	3.0	3.0	3.0	100.0	4.0	200.0	200.0	500.0
La	30.0	100.0	100.0	300.0	25.0	1000.0	1000.0	5000.0
Ce	30.0	100.0	100.0	300.0	1.0	1000.0	1000.0	4000.0
Pr	100.0	1000.0	1000.0	1000.0	25.0	1000.0	1000.0	5000.0
Nd	100.0	1000.0	1000.0	1000.0	25.0	1000.0	1000.0	5000.0
Pm	100.0	1000.0	1000.0	1000.0	25.0	1000.0	1000.0	5000.0
Sm	100.0	1000.0	1000.0	1000.0	25.0	1000.0	1000.0	5000.0
Eu	100.0	1000.0	1000.0	1000.0	25.0	1000.0	1000.0	5000.0
U	10.0	10.0	10.0	67.0	2.0	60.0	60.0	.5
Np	10.0	10.0	10.0	6.0	10.0	400.0	400.0	300.0
Pu	3.0	200.0	200.0	1000.0	3.5	100.0	100.0	350.0
Am	25.0	1000.0	1000.0	5000.0	25.0	1000.0	1000.0	5000.0
Cm	25.0	1000.0	1000.0	5000.0	25.0	1000.0	1000.0	5000.0

[a] Concentration factors for crustaceans, mollusks and finfish are for edible portions only.

Table D7. Stable Element Metabolic Half-Lives and Associated Metabolic Decay Rates for Finfish[9, (a)]

Elements	Metabolic Half-life (days)	Metabolic Decay Rate (yr^{-1})[(b)]
H	1−100[(c)]	2.5+0
Co	100−1000	2.5−1
Ni	100−1000[(d)]	2.5−1
Rb	1−100[(d)]	2.5+0
Sr	100−1000 and more	2.5−1
Y	100−1000[(c)]	2.5−1
Zr	100−1000[(d)]	2.5−1
Nb	100−1000[(d)]	2.5−1
Mo	1−10[(d)]	2.5+1
Tc	1−10[(d)]	2.5+1
Ru	1−100[(d)]	2.5+0
Rh	1−100[(d)]	2.5+0
Pd	10−100[(d)]	2.5+0
Ag	1−100[(d)]	2.5+0
Cd	100−1000[(d)]	2.5−1
Sn	10−100[(d)]	2.5+0
Sb	10−100[(d)]	2.5+0
Te	10−100[(d)]	2.5+0
I	1−100	2.5+0
Cs	10−1000	2.5−1
Ba	100−1000[(c)]	2.5−1
La	100−1000[(d)]	2.5−1
Ce	100−1000[(d)]	2.5−1
Pr	100−1000[(d)]	2.5−1
Nd	100−1000[(d)]	2.5−1
Pm	100−1000[(d)]	2.5−1
Sm	100−1000[(d)]	2.5−1
Eu	100−1000[(d)]	2.5−1
U	100−1000[(d)]	2.5−1
Np	>1000[(d)]	2.5−1
Pu	>1000[(d)]	2.5−1
Am	>1000[(d)]	2.5−1
Cm	>1000[(d)]	2.5−1

[a] In view of the relative scarcity of metabolic half-life data for shellfish,[9] the half-lives in these organisms are assumed to be the same as those given for finfish.

[b] Estimated as minimum possible from metabolic half-life given.

[c] Estimated from consideration of the behavior of closely-related elements.

[d] Taken from data for small terrestrial vertebrates.

D2.1.2.3 Irrigated Food

The pathway involving irrigated food consists of three major portions:[1,3] irrigated foods ingested directly by the human population; irrigated food utilized as foods for the animal population, products and by-products of which are, in turn, used as food for the human population; and "irrigation water" used as drinking water for animals, derived products of which are then used for human consumption.

The dose received by the population from ingestion of irrigated foods and derived animal products between times 0 and t can be written as

$$\text{dose}_a = \sum_m U_i^m \sum_i d_{ii} f_{im}\, e^{-\lambda_i t_m} \int_o^t C_{im}(t')\, dt'$$
$$+ \sum_m U_i^{m'} \sum_i d_{ii} f_{im'}\, e^{-\lambda_i t_{m'}} \int_o^t C_{im'}(t')\, dt', \quad \text{(D2-8)}$$

where

U_i^m is the individual consumption rate for irrigated food m,
f_{im} is the processing or removal factor for nuclide i and food m,
t_m is the delay time between harvesting and ingestion of m,
$C_{im}(t)$ is the concentration of radionuclide i in ingested food m, and
$C_{im'}(t)$ is the concentration of radionuclide i in animal byproduct or product m'.

The concentration of radionuclide i in any irrigated crop m can be written as[1,3]

$$C_{im}(t) = I\,\{f_{im}^{s} C_{im}^{s'}(t) + f_{im}^{r} C_{im}^{r'}\}, \quad \text{(D2-9)}$$

where

$$C_{im}^{s'}(t') = e^{-\lambda_{ei} t'} \int_o^{t'} C_{iw}(t)\, e^{+\lambda_{ei} t}\, dt;$$

$$C_{im}^{r'}(t') = e^{-\lambda_{ri} t'} \int_o^{t'} C_{iw}(t)\, e^{+\lambda_{ri} t}\, dt;$$

$$f_{im}^{s} = (r f^{s} T_m / Y_v)\,;$$

$$f_{im}^{r} = (f B_{im} / P). \quad \text{(D2-10)}$$

In these expressions, the indices s and r denote sprinkling and root uptake, respectively,

I is the rate at which irrigation water is applied per unit area,

λ_{ei} is the effective removal rate constant for sprinkling with $\lambda_{ei} = \lambda_i + \lambda_w$, where λ_w is the weathering rate constantfor sprinkling,

λ_{ri} is the effective removal rate for root uptake with $\lambda_{ri} = \lambda_i + \lambda_r$, where λ_r is the uptake rate constant,

f^s is the fraction of irrigation water applied by sprinkling,

r is the fraction of activity retained on the crop,

T_m is the fraction of the activity obtained from sprinkling deposition that ends up in the edible portion of crop m,

Y_v is the agricultural productivity in wet weight/area,

f is the fraction of the year that crops are irrigated,

B_{im} is the concentration factor for root uptake in (activity/wet weight of crop) per (activity/dry weight of soil),

P is the "effective surface density" of the soil.

$C^{s'}_{im}(t)$ is proportional to the buildup of activity on the crop with time due to repeated sprinkling and

$C^{r'}_{im}(t)$ is proportional to the buildup in the soil due to continued irrigation.

Similarly, the concentration of radionuclide i in any animal product m′ can be written as[1,3]

$$C_{im'}(t) = f'_{im'}\, I \{f^s_{im} C^{s'}_{im}(t) + f^r_{im} C^{r'}_{im}(t)\} + f'_{iw}\, C_{iw}(t), \qquad \text{(D2-11)}$$

where

f'_{im} is the product of the transfer coefficient of the animal product[1,3] and the consumption rate of the animal.

The transfer coefficient is a rough approximation to the ratio of the amount of radionuclide in the animal product under consideration to the amount of radionuclide ingested.

Realistic ranges and average values for the delay times are given in Table D8.[10] The values for the irrigated-food concentration factors, $\{B_{im}\}$, and for the animal product transfer coefficients are given in Table D9.[11] The quantities assumed for the daily ingestion rates for animals are presented in Table D10.[10] The values used for all the other parameters for the irrigation model are listed in Table D11.[1,3,11]

Table D8. Irrigated Food Delay Times

Time Period	Delay Time*
Time between removal of water from waterbody and use of water for irrigation	0 to days (0 days)
Time between application of irrigation water and harvest	Days to years (2 months)
Time between harvest and human consumption	Days to years (120 days)
Time between harvest and animal consumption	Days to years (90 days)
Time between animal consumption and slaughter	Days to years (15 months)
Time between poultry consumption and slaughter	Days to months (3 months)
Time between slaughter and consumption	Days to months (2 months)
Time between milking and consumption	Days to months (40 days)
Time between laying and consumption	Days to months (27 days)

*Based on figures presented in *Agricultural Statistics 1974*.[10]

Table D9. Plant Concentration Factors and Animal Product Transfer Coefficients[11]

	B_{im}	S_{ia}				
Element	Plant/Soil (Dimensionless)	Egg/Feed (day/kg)	Milk/Grass (day/kg)	Beef/Feed (day/kg)	Pork/Feed (day/kg)	Poultry/Feed (day/kg)
Co	9.4E−03	1.0E−01	5.0E−04	1.0E−−03	5.0E−03	1.0E−03
Ni	1.9E−02	1.0E−01	3.4E−03	1.0E−03	5.0E−03	1.0E−03
Rb	1.3E−01	3.0E−00	1.0E−02	1.5E−01	2.0E−01	2.0E+00
Sr	2.0E−01	4.0E−01	1.5E−03	3.0E−04	7.3E−03	9.0E−04
Y	2.5E−03	5.0E−04	5.0E−05	5.0E−03	5.0E−03	5.0E−04
Zr	1.7E−04	1.2E−03	2.5E−04	5.0E−04	1.0E−03	1.0E−04
Nb	9.4E−03	1.2E−03	1.2E−03	5.0E−04	1.0E−03	1.0E−04
Mo	1.3E−03	4.0E−01	4.0E−03	1.0E−02	2.0E−02	2.0E−03
Tc	2.5E−01	9.9E−04*	1.2E−02	9.9E−04	9.9E−04	9.9E−04
Ru	1.0E−02	4.0E−03	5.0E−07	1.0E−03	5.0E−03	3.0E−04
Rh	1.3E+01	4.0E−03	5.0E−03	1.0E−03	5.0E−03	3.0E−04
Pd	5.0E+00	4.0E−03	5.0E−03	1.0E−03	5.0E−03	3.0E−04
Ag	1.5E−01	9.9E−04	2.5E−02	9.9E−04	9.9E−04	9.9E−04
Cd	3.0E−01	9.9E−04	6.2E−05	1.6E−02	1.6E−02	1.6E−02
Sn	2.5E−03	9.9E−04	1.3E−03	9.9E−04	9.9E−04	9.9E−04
Sb	1.1E−02	7.0E−02	7.5E−04	3.0E−03	7.0E−03	6.0E−03
Te	1.3E+00	4.0E−01	5.0E−04	5.0E−02	1.0E−02	1.0E−02
I	2.0E−02	1.6E+00	1.0E−02	2.0E−02	9.0E−02	4.0E−03
Cs	2.0E−02	6.0E−01	5.0E−03	3.0E−02	2.6E−01	4.5E+00
Ba	5.0E−03	4.0E−01	4.0E−04	5.0E−04	1.0E−02	5.0E−04
La	2.5E−03	2.0E−03	2.5E−06	5.0E−03	5.0E−03	4.0E−03
Ce	5.0E−04	3.0E−03	1.0E−05	1.0E−03	5.0E−03	6.0E−04
Pr	2.4E−03	4.0E−03	2.5E−06	5.0E−03	5.0E−03	1.0E−03
Nd	2.5E−03	2.0E−04	2.5E−06	5.0E−03	5.0E−03	4.0E−03
Pm	2.5E−03	7.0E−03	2.5E−06	5.0E−03	5.0E−03	1.0E−04
Sm	2.5E−03	7.0E−03	2.5E−06	5.0E−03	5.0E−03	4.0E−03
Eu	2.5E−03	7.0E−03	2.5E−06	5.0E−03	5.0E−03	4.0E−03
U	2.5E−03	3.4E−01	6.0E−04	5.0E−03	6.0E−01	1.2E−03
Np	2.5E−03	2.0E−03	2.5E−06	5.0E−03	1.0E−02	4.0E−03
Pu	2.5E−04	2.0E−03	2.5E−08	5.0E−03	1.0E−02	4.0E−03
Am	2.5E−04	2.0E−03	2.5E−06	5.0E−03	1.0E−02	4.0E−03
Cm	2.5E−03	2.0E−03	2.5E−06	5.0E−03	1.0E−02	4.0E−03

*Where value unknown, a default value of 9.9E−04 was used.

Table D10. Consumption Rates of Feed and Water by Farm Animals[11]

Farm Animal	Feed or Forage (kg/day)	Water (ℓ/day)
Milk Cow	55 (fresh forage)	60
Beef Cattle	68 (dry feed)	50
Pig	4.2 (dry feed)	10
Poultry (chickens)	0.12 (dry feed)	0.3

Table D11. Miscellaneous Irrigation Pathway Parameters

Symbol	Interpretation	Value[1,3,11]
λ_w	Weathering rate constant	18.07 yr^{-1} (corresponds to a 14 day weathering half-life)
λ_r	Root uptake rate constant	$(f\ B_{im}/P) = .0028\ B_{im}\ yr^{-1}$
f^s	Sprinkling fraction	See Appendix E, Table E44
r	Fraction retained	.25
Y_v	Agricultural yield	2.0 kg/m² for leafy vegetables or produce ingested by humans
f	Fraction of year for irrigation	2/3, assuming 2 crops per year and 120 days per crop
T_m	Translocation fraction for crop m	1.0 for leafy vegetables; 0.1 for all other crops
P	Effective "surface density"	240 kg/m²
t_m	Delay time for crop m	.038 yr (14 days)

D2.1.2.4 Shoreline Usage

The shoreline pathway involves external exposure to contaminated shorelines and floodplains. The function representing the shoreline pathway contribution to the dose to an individual from exposure between times 0 and t can be written as

$$\text{dose}_s = U_s \sum_i d'_{is}\, f_{is} \int_o^t C_{is}(t')\, dt' \, , \qquad \text{(D2-12)}$$

where

U_s is the individual shoreline usage rate,
d'_{is} is the modified shoreline dose factor for nuclide i,
f_{is} is the shoreline width factor, and
$C_{is}(t)$ is the shoreline sediment concentration of nuclide i.

The modified shoreline dose factor is the product of the regular shoreline dose factor and the effective surface density of the soil.[13] The effective surface density of the soil is taken to be 40 kg/m²; this corresponds to the assumption that all the radioactivity contained in the top 2.5 cm of sediment effectively resides on the shoreline surface. The shoreline width factors are taken to have the values given in Table D12.[4,12]

The primary difference between the procedure for obtaining the shoreline dose used in this study and that recommended in Regulatory Guide 1.109 is in the calculation of the concentration of radioactivity in the shoreline sediment; here, time-dependent buildup is permitted and some environmental weathering is included. In particular, in this study, the concentration of nuclide i in the shoreline sediment is assumed to be represented by

$$C_{is}(t') = K\,A\,e^{-(\lambda_i+\alpha)t'} \int_o^{t'} C_{iw}(t)\, e^{(\lambda_i+\alpha)t} dt$$

$$+ K\,B\,e^{-(\lambda_i+\beta)t'} \int_o^{t'} C_{iw}(t)\, e^{(\lambda_i+\beta)t} dt \, , \qquad \text{(D2-13)}$$

where

K is the water to sediment transfer constant,
λ_i is the decay constant of nuclide i,
$C_{iw}(t)$ is the concentration of nuclide i in the water adjacent to the sediment,

and the remaining parameters, (α, β, A and B), which are related to weathering, are as described below.

Table D12. Shoreline Width Factors

River	0.2
Lake	0.3
Ocean	0.5
Estuary	1.0

Experimental evaluation on the Columbia River indicates that K for a river is approximately independent of radionuclide species and has a value of about 631 ℓ/kg-yr.[3,13,14] In this report, this value is used for all waterbodies, even though a lower rate would probably be appropriate for large lakes and oceans.

In this study, the treatment of environmental removal parallels that used in the Reactor Safety Study (RSS).[15] The weathering function W(t) is taken to be the sum of two exponentials and approximately represents the fraction of deposited radioactive material remaining after time t. It can be written as

$$W(t) = Ae^{-\alpha t} + Be^{-\beta t} , \qquad \text{(D2-14)}$$

where

A represents the fraction of the radioactive material with the comparatively short weathering decay constant α,

B represents the fraction of the radioactive material with the relatively long weathering decay constant β, and

$$A, B \geq 0,$$
$$A + B = 1.$$

The selection of the parameter values used in the weathering function reflects the kinds of sediments typically associated with each shore line type and their respective grain sizes (see Appendix E). It is assumed that the soils of the floodplains of the river and the lake are similar to the soil types of the RSS.[15] The ocean beach is taken to be composed of sorted sands larger than 100 μ.[16] The estuary shoreline is assumed to be characterized by fine-grained sand and clay particles smaller than 100 μ.[16] The weathering parameters assumed are presented in Table D13.

D13. Shoreline Weathering Parameters[15,16]

Shoreline Type	A	τ_α	B	τ_β
Lake	.63	180 days	.37	90 years
Ocean	.90	15 days	.10	180 days
River	.63	180 days	.37	90 years
Estuary	.05	180 days	.95	90 years

In that table, τ_α is the weathering half-life for fraction A, and τ_β is the weathering half-life for fraction B. As expected, for the parameters employed the impact of weathering on both the shore line concentrations and the resultant individual doses is slight for an estuary, but significant for a river, lake or ocean shore-line.

D2.1.2.5 Immersion

The pathway involving immersion (bathing) is taken to consist entirely of exposure due to contaminated natural water. That is, immersion due to usage of artificial pools and that due to actual bathing are both taken to occur in untreated water. As a result, the dose received by an individual for exposure between time 0 and t from swimming and other immersion can be written as

$$\text{dose}_b = U_b \sum_i d_{ib} \int_0^t C_{iw}(t')\, dt' , \qquad \text{(D2-15)}$$

where

U_b is the individual immersion rate, and
d_{ib} is the immersion dose factor for isotope i.

D2.2 Population Dose

D2.2.1 General Expression

As was already mentioned, all the individuals in a pathway do not receive the same exposure. Differences in exposure can be caused both by differences in individual usage patterns and by spatial and temporal variations in radionuclide concentrations. Consequently, it is reasonable to regard a population as being composed of subgroups differing either in usage patterns or in concentrations experienced in the pathway or in both.

As a result, the dose received by a population from exposure to pathway p can be written as

$$\text{dose}_p = \sum_{p'} P_{pp'} U_{p'} \sum_i d_{ip}\, f_{ip}\, e^{-\lambda_i t_p} \int_0^t C_{ipp'}(t')\, dt' , \qquad \text{(D2-16)}$$

where

$P_{pp'}$ is the population size of subgroup p′ in pathway p,

and all other symbols are as previously defined. The values used for the population sizes and the usage rates are presented in Appendix E.

D2.2.1.1 Variations Among Individuals

The general expression for the dose to the population, Equation (D2-16), takes into account the fact that the individuals within the population can have rather different usage habits for any pathway. To be rigorous, the set of individual usage habits should be regarded as forming a distribution function of usage. However, determination of such distribution functions is beyond the scope of this study. A uniform usage or exposure rate is assumed for all members of a population for each pathway. The resulting impact of this assumption on the calculated population doses should be negligible. (In contrast, the effect on the resultant health effects could be substantial.)

The general expression given in Equation (D2-16) also does not take into account the fact that individuals can be affected quite differently by exposure to the same amount of radioactivity. Differences result from age, metabolic behavior, size and many other factors. Such effects are not necessarily insignificant. However, the uncertainties in both the estimation of doses and the descriptions of the exposed populations as functions of such factors probably preclude any reasonable treatment of these factors at this time. Therefore, such differences are ignored in this study.

D2.2.1.2 Population Changes

The population exposure for any pathway can be expected to change over an extended period for many reasons. This study assumes that the product of the population size and the individual usage rate remains constant over the entire time period under consideration. Utilizing detailed population usage trends would generally not change the calculated results for any pathway at any given site by more than a factor of 2.

D2.2.1.3 Location

In many waterbodies, the concentration in the water is a strong function of location. For example, in a river the concentration can be expected to change at least at each point of dilution. Consequently, the dose to the population must take this dependence into account. This dependence is considered in more detail in the following sub sections.

D2.2.2 Specific Pathway Expressions

D2.2.2.1 Dry Site

Only two potentially major exposure pathways exist at a dry site: drinking water and irrigation. The concentration of a radionuclide in an aquifer, and therefore the dose resulting from either pathway, is a strong function of location. However, inasmuch as a generic site is being discussed, it is probably not necessary to consider usage via withdrawal from discrete locations such as wells; instead, usage can probably be taken to be distributed uniformly over the extent of the aquifer without causing serious error. (Indeed such a treatment is probably preferable for a generic study inasmuch as it tends to eliminate artifactual results due to the arbitrary particular choices of a few well locations.)

The dose to the population from drinking water taken from an aquifer and consumed between times 0 and t can be written as

$$\text{dose}_w = P^a_w\, U_w \sum_i d_{ii} e^{-\lambda_i t_w} \iint\limits_{\substack{\text{area of}\\ \text{aquifer}}} \int_0^t C_{id}(x,y;t')\, dt'\, dx\, dy\ , \qquad \text{(D2-17)}$$

where

P^a_w is the water drinking population per unit area of the aquifer,

and all other terms are as defined previously.

The population dose from the irrigation pathway can be treated in a similar manner with respect to its spatial characteristics. Unlike the drinking water population, the irrigated food population is not necessarily the population residing in the area under consideration; instead, it is the population that ingests the food grown in that area, taken in proportion to the yield of irrigated food from that area.

D2.2.2.2 Lake

All five of the main exposure pathways can exist at a lake. Due to the mixed-tank approximation employed in the modeling of lakes, the concentration is essentially independent of the location within the lake.* As a result, the population dose for any pathway at a single lake ignores the locations of the members of the population. However, if a chain of lakes is considered, then the concentration is a function of the location in the chain.

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*This assumes that the contribution of the nearshore region can be ignored.

For such a chain of lakes, the dose to the populations for water consumed between times 0 and t can be written as

$$\text{dose}_w = \sum_n P^1_w(n)\, U_w \sum_i e^{-\lambda_i t_w} \int_o^t C_{id}(n;t')\, dt' , \qquad \text{(D2-18)}$$

where

$P^1_w(n)$ is the drinking water population for the nth lake, and
$C_{id}(n;t)$ is the concentration of isotope i in the drinking water taken from the nth lake.

The population doses for the other pathways can be treated in analogous manners with respect to their spatial distributions. The appropriate aquatic food and irrigated food populations are not necessarily the populations residing around the lakes; instead they consist of the people who consume the foods contaminated by the lake water.

D2.2.2.3 Estuary

Only three of the exposure pathways considered in this study exist at an estuary site:* aquatic food ingestion, shoreline usage and immersion. In the modeling of estuaries in this report, the concentration of a radionuclide is taken to be a function only of distance along the estuary.

The aquatic foods are assumed to be harvested uniformly over the area of the estuary. Consequently, the population dose from investion of aquatic food taken from an estuary can be written as

$$\text{dose}_a = \sum_m P^{em}_a U^m_a \sum_i d_{ii}\, f_{im}\, e^{-\lambda_i t_m} \int_{\substack{\text{length}\\ \text{of}\\ \text{estuary}}} \int_o^t C_{im}(x;t')dt\, dx , \qquad \text{(D2-19)}$$

where

p^{em}_a is the ingestion population per unit length of estuary for species m, and
$C_{im}(x;t')$ is the concentration of isotope i in aquatic food m which is located initially at a distance x from the source of radioactivity.

The doses due to shoreline usage and immersion in water can be treated in similar manners if analogous spatial distributions of usage are assumed.

D2.2.2.4 Ocean

The same three pathways are found at an ocean site as are present at an estuary site: aquatic food ingestion, shoreline usage, and immersion. Near the coast, the concentration of a radionuclide is taken to be a function of both the distance along the shore and the distance from the shore.

The aquatic foods are assumed to be harvested uniformly along the shoreline; furthermore they are taken to be harvested uniformly out to some specified offshore distance. Consequently, the population dose from ingestion of aquatic food taken from the ocean can be written as

$$\text{dose}_a = \sum_m P^{om}_a U^m_a \sum_i d_{ii} f_{im}\, e^{-\lambda_i t_m} \cdot \iint_{\substack{\text{region}\\ \text{of}\\ \text{harvest}}} \int_o^t C_{im}(x,y;t')\, dt'\, dx\, dy , \qquad \text{(D2-20)}$$

where

P^{om}_a is the aquatic food ingestion population per unit area of harvest region for species m, and
$C_{im}(x,y;t')$ is the concentration of radionuclide i in aquatic food m, which, prior to harvest, is located at (x,y) with respect to the accident site.

In contrast, the shoreline users are assumed to be uniformly distributed along the coast. Therefore, the population dose due to shoreline exposure along an ocean coast can be written as

$$\text{dose}_s = P^o_s U^o_s \sum_i d_{is} \int_{\substack{\text{length}\\ \text{of}\\ \text{coast}}} \int_o^t C_{is}(x,0;t')\, dt' dx , \qquad \text{(D2-21)}$$

where

P^o_s is the shoreline population per unit length of ocean coast,
P^o_b swimming population per unit length, and
$C_{is}(x,0,t')$ is the concentration of radionuclide i in the sediment, which is located at (x,0) with respect to the accident site.

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*In parts of some estuaries, the water can be used for irrigation. However, this use is ignored here.

The dose due to immersion can be obtained by a similar treatment.

D2.2.2.5 River

All five exposure pathways can exist along a river system. In any given system, the concentration of a radionuclide at any point is usually primarily a function of distance downstream. In the modeling of rivers in this study, any river system is divided into a series of compartments, in each of which the concentration of the activity is effectively taken to be a constant. For long river systems, the populations can probably be considered to be uniformly distributed along large stretches of the system but probably not over the entire length. Consequently, the river is divided into sections over which the various populations are assumed to be uniformly distributed and the concentration is taken to be constant.

As a result, the drinking water dose to the population along a river system can be written as

$$\mathrm{dose}_w = \sum_n P^r_w(n)\, U_w \sum_i d_{i1}\, e^{-\lambda_i t_w} \int_0^t C_{id}(n;t')\, dt'\, \Delta x_n \ , \qquad \text{(D2-22)}$$

where

$P^r_w(n)$ is the drinking water population per length along the nth section of the river system, and
Δx_n is the length of the nth section.

It can be seen that this is almost the same expression as that used to describe the dose to the population for a series of lakes. The expression for the drinking water population dose in a river system is obtained if the following substitution is made in the formula for the dose at a lake:

$$P^l_w(n) \rightarrow P^r_w(n)\, \Delta x_n \ .$$

Analogous changes have been used to obtain all the expressions for the population doses along river systems for all the other pathways.

D2.3 Pathway Model Limitations

The limitations in the pathway considerations are primarily of two types: weaknesses in the models and difficulties in obtaining appropriate values for parameters in the models. The primary sources of weaknesses in the models are the inadequate inclusion of the time dependence of the movement of the radionuclides within the environment and the lack of consideration of the interconnections between the various portions of the environment. Only the food-web model mentioned earlier, and described in detail in Appendix C, and in SAND80-1670, attempts to treat both the time dependence and the interconnections in a more appropriate fashion. (Corresponding models could be developed both for the irrigation pathway and for the shoreline exposure pathway.* Such models would consider the mechanisms by which the radioactivity passed through various portions of the environment. Thus the improved models would permit a better description of the buildup of radioactivity in the pathways as functions of both time and various environmental conditions.)

Another source of weakness in some of the models is the failure to include conservation of the radionuclide mass. In general, in the models the radioactivity is not transferred from the waterbody to a given pathway; instead, that radioactivity is counted as being in both places simultaneously.

The difficulties in obtaining adequate values for the parameters in the pathway models are due, in part, to the large number of parameters required. In addition, most of the parameters used in the pathway models are relatively site-dependent. It would not be easy to obtain all the required parameters for a specific site. It is even more difficult to obtain meaningful values for parameters for a generic study such as this one. Unfortunately, the resultant doses depend critically on some of these parameters.

In particular, certain element-dependent parameters such as concentration factors and bioloical elimination rates are subject to many sources of error. They tend to be extremely location- and species-dependent. Furthermore, they tend to be difficult to obtain experimentally even for specific sites and species. The generic values employed in this study may not even fall within an order of magnitude of the appropriate values. In addition, their basic applicability depends on the adequacy of certain first-order approximations used in the models employing them.

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*The pathway model which is of potential importance for this study but which is least adequate is the shoreline usage model. The accumulation of radioactivity on shorelines and floodplains involves both deposition of contaminated sediment and contamination of previously-deposited sediment. Relevant processes include flooding, tidal movements, wave actions, and weathering processes. The amount of accumulation is expected to be a strong function of both these processes and the composition of the involved sediment particles. The removal of radioactivity from the shore-lines and floodplains is likewise very complicated. The current model includes few of the relevant factors affecting deposition and removal. Consequently, any correspondence its results have to reality may be fortuitous. The improvements made in the shoreline model[1] for this study constitute a preliminary attempt to remedy some of the inadequacies.

Another source of uncertainty in parameters such as concentration factors arises because of the tendency of certain radionuclides to concentrate in specific organs in aquatic organisms and irrigated plants; such localization can result in either a much higher or a much lower than average concentration in the portions normally ingested by humans. Furthermore, no account is taken of the fact radionuclides from different foods are not assimilated equally well by humans.

Uniform population usage rates have been assumed for all pathways. In reality, these could vary significantly with the passage of time. Other problems related to the population sizes and the associated usage rates are discussed in Appendix E.

D2.4 Dose Factors

D2.4.1 Internal Dose Factors

In this study, ingestion is the only internal exposure mechanism considered. Two different sets of ingestion dose factors were utilized in the calculations: one generated by INREM II[17-19] and one generated by the dosimetry codes employed in the Reactor Safety Study (RSS).[15] The values of the former set of dose factors are listed in References 18 and 19; the values of the latter were generated at ORNL[20] for this study and are listed in Addendum D3 of this appendix. The more complete INREM II set was used for determining the list of important isotopes; the RSS set was used for calculating the final results. As in the RSS, only adult dose factors were used in estimating doses for the consideration of resultant health effects.

D2.4.2 External Dose Factors

In this study, two different types of external exposure are considered: exposure by immersion in contaminated water and exposure to contaminated shorelines and floodplains. The dose factors for both of these types of exposure were taken from the "Year 2000 Study" (YTTS).[3,5] Although the dose factors in that study were determined somewhat differently than those used in the RSS, this is not thought to be important in limiting the comparison of the results of this study to those of the RSS.

The immersion pathway contributed very little in this study to the total impact at any site. Consequently, even if the immersion dose factors employed were off by an order of magnitude, the total impact of that pathway would still not be significant relative to that of other pathways.

Although the shoreline dose factors in the two studies appear to be extremely different, the YTTS dose factors and the RSS dose factors should produce approximately equivalent results if used appropriately. In the RSS, the dose factors for exposure to a contaminated plane contained no roughness factor, whereas those generated for the other study contained a roughness factor of 0.5. However, an analogous shielding factor of 0.33 or 0.50, depending on conditions, was included elsewhere in the RSS calculations. Furthermore, the RSS shoreline dose factors were obtained without considering the penetrating power of the radiation whereas the YTTS dose factors took such considerations into account. As a result, the YTTS shoreline dose factors for the skin correspond more closely to those designated as total-body dose factors in the RSS.* Consequently, the YTTS shoreline skin dose factors have been utilized in this study as the dose factors both for the total-body and for all the internal organs.

D3 Health Effects

D3.1 Introduction

Health effects models translate the radiation dose received by the human population into the resultant health effects such as cancer fatalities, thyroid effects and genetic mutations. The health effects models used in this study are essentially identical to those used in the RSS,[15] except where noted otherwise.

D3.2 Acute Effects

D3.2.1 Introduction

In the RSS, health effects were divided into acute, latent and chronic. The acute effects were due primarily to large early total-body doses of 100 rads or more to individuals. These effects included both early morbidities and mortalities occurring within a year after the exposure. The acute effects considered were all somatic and included direct radiation damage to certain organs, thyroid impairment, prenatal and neonatal deaths, and sterility.

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*In the former study, if the radiation had low penetrating power, then it contributed primarily to skin dose and not to the total-body dose. Radiation which could penetrate 7×10^{-3} cm of tissue was considered to contribute to the skin dose; irradiation which could penetrate 5 cm of tissue also contributed to the total-body dose and to the doses of other organs than skin. In the calculation of the RSS shoreline dose factors, penetrating power was not considered.

A dose was considered to be acute only if the necessary dose occurred over a short enough period of time. Inasmuch as the greatest atmospheric exposure would occur immediately after the accident, the calculation of acute doses in the RSS was done in the period right after the accident. The relevant time periods ranged from 2 days to 1 year, depending upon both the organ and the health effect being considered.

The primary organs for which acute effects were calculated were bone marrow, lung and gastrointestinal tract. Effects were also considered for the thyroid, the testes and the ovaries.

D3.2.2 Estimation of Acute Effects

For acute effects, dose-response curves were employed to convert individual doses to health effects. Such curves plot the individual dose versus the probability of the effect. Each of these curves is characterized by a low dose threshold below which no effect occurs and a high dose threshold above which the effect always occurs (Figure D1).

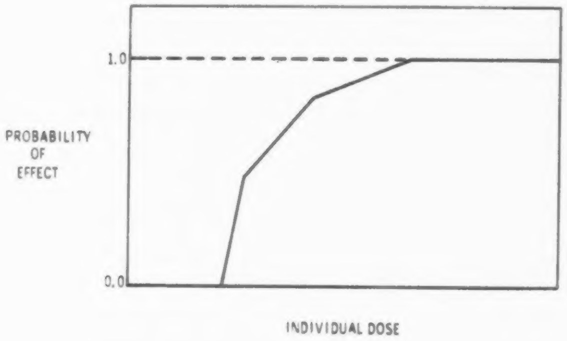

Figure D1. Dose-Response Curve for Acute Effects to an Individual

The number of acute effects of type j can be written as

$$(\text{health effect})_j = \sum_{p'} P_{p'} \cdot \beta_j(\text{dose}_{p'}) \ , \qquad \text{(D3-1)}$$

where

$P_{p'}$ is the population for subgroup p′, and
$\beta_j(\text{dose}_{p'})$ is the probability of effect j for total individual dose p′.

Each total individual dose is the sum of the contributions from all possible pathways, {p}. That is,

$$\text{dose}_{p'} = \sum_{P} (\text{individual dose})_{pp'} \ . \qquad \text{(D3-2)}$$

Unlike the atmospheric release considered in the RSS, the entire hydrospheric release is generally expected to be rather slow. Consequently, the radiation exposure of the human population via hydrospheric pathways is usually expected to be over a relatively protracted period of time at a rather low rate. In general, no acute effects are expected. Typically, very large doses of radiation to individuals would be possible only through continued drinking water ingestion at certain aquifer sites; the concentration of radionuclides in the water in the aquifers could be relatively high so that large amounts of radioactivity could be received by individuals within a period on the order of months.

A check for potentially acute effects is done in this study over all one-year periods. Thus, the possibility exists of over-estimating certain acute effects relative to the number which would have been predicted using the shorter RSS accumulation times for acute effects for some organs. However, acute doses are not thought to be likely because of the interdictive procedures which could easily be taken to prevent such large doses from the liquid pathways. Therefore, the error is thought to be of negligible importance.

D3.3 Latent and Chronic Effects

D3.3.1 Introduction

In the RSS, the latent and the chronic effects were considered to be random phenomena based on the total doses to the entire population. In general, the doses to individuals which resulted in latent and chronic effects were much less than those which caused acute effects. The effects included cancers, genetic effects and benign and malignant thyroid nodules, all occurring many years after the exposure. As was indicated previously, most, if not all, of the effects obtained through liquid pathways are expected to be chronic.

The eight categories of "organs" and the associated health problems considered for latent and chronic effects in the RSS are as follows:

bone marrow	leukemia
lung	lung cancer
skeletal bone	bone cancer
"other"	breast cancer

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lower large intestine	gastrointestinal tract cancer
"other"	pancreas and all other cancer not mentioned above, excluding thyroid cancer
whole body	all cancers, except thyroid
thyroid	benign nodules, malignant nodules and hypothyroidism
testes	genetic effects

As in the RSS, the associated health effects for the first six "organs"* were summed to obtain the total number of chronic cancer mortal ities. Alternatively, the whole-body dose was calculated and used to estimate the total number of nonthyroid cancer fatalities. Current cure rates were included in the determination of the number of cancer fatalities. Thyroid and genetic effects were always estimated separately.

D3.3.2 Estimation of Latent and Chronic Effects

In general, the number of latent or chronic effects of type j can be written as

$$
\begin{aligned}
(\text{health effect})_j = h_j \cdot \sum_{p'} P_{p'}\ \delta_j\,(\text{dose}_{p'}) \\
\sum_{p} (\text{individual dose})_{pp'}\ , \qquad \text{(D3-3)}
\end{aligned}
$$

where

h_j is the health effects factor for effect j,
$P_{p'}$ is the population for subgroup p', and
$\delta_j(\text{dose}_{p'})$ is the dose effectiveness factor for individual p'.

The various latent and chronic effects models differ in the values assumed for the health effects factors and the dose effectiveness factors.

D3.3.2.1 Cancer

For latent and chronic cancer, an absolute-risk model was employed in the RSS. That is, the risk of any cancer was assumed to be constant over a given finite plateau period; no effects occur before or after the plateau period. A relative-risk model in which the risk of cancer increases throughout an individual's lifetime was not used.

Three estimates were made of the latent and chronic cancer fatalities for all organs other than the thyroid: an upper bound, a central estimate, and a lower bound. The upper bound was based on the BEIR or linear, no-threshold hypothesis (Figure D2a). In that model, it is assumed that the number of health effects varies directly with the population dose. Health effects factors (expected fatalities/population dose) are used as the proportionality constants (Table D14). No individual dose threshold is employed.

The central estimate is based, in part, on the BEIR hypothesis but assumes that doses administered at lower dose rates or in lower total individual doses are less effective in causing health effects (Figure D2b). Dose effectiveness factors (Table D15) are used to modify the effects calculated for individual low dose rates and/or low total doses. The only cancers excepted from the use of dose effectiveness factors are breast and thyroid. In the RSS, the appropriate dose effectiveness factors for the central estimate for all time were based on the first-year early dose; no explicit regard was given to the rate at which the dose was received. In effect, only the diagonal elements of Table D15 were utilized.

The lower estimate assumes that doses received at very low dose rates and/or in very low individual doses are not effective at all in causing health effects (Figure D2c). Consequently, an individual threshold dose rate or threshold total dose is assumed, below which no effect is considered to occur.

In the atmospheric release considered in the RSS, the dose is delivered at its highest rate immediately after the accident; that rate tapers off approximately exponentially with the passage of time. Therefore, it is conservative to use the first-year doses to estimate the dose-effectiveness factors. The hydrospheric releases considered in this study are usually delivered with much different temporal distributions than an atmospheric release. It can take years after an accident for any radioactivity to contact the human population via hydrospheric pathways. The rate at which the radiation reaches the population will generally be rather low; the time period over which exposure occurs will be relatively long. The doses and time periods which should be used to determine dose effectiveness factors are not as readily defined as they were for atmospheric releases.

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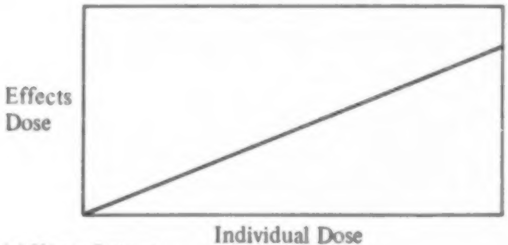
*In the RSS, the dose to "other" was taken to be the average dose to all the organs considered other than marrow, lung, skeletal bone, lower large intestine, thyroid, testes, endosteal cells, stomach wall, small intestine, upper large intestine and ovaries. The dose to any of the other organs is estimated to be approximately the same for a given isotope. Consequently, in this study, the dose to the organ designated as "other" was taken to be that to the pancreas. If the dose to the pancreas was not available, then the dose to "other" was taken to be the dose to the whole body.

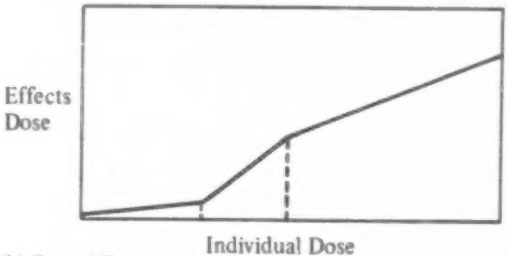
Table D14. Expected Total Latent Cancer (excluding thyroid) Deaths per 10^6 Person-rem[15]

	Expected Cancer Deaths Per 10^6 Person-rem[(a)]									
	From Internal Sources									From External Sources
	Postaccident Period Over Which Dose is Accumulated									
Type of Cancer	0–1	1–10	11–20	21–30	31–40	41–50	51–60	61–70	71–80	
Leukemia	28.4	26.0	18.7	13.8	9.7	6.8	4.0	1.7	0.5	28.4
Lung	22.2	22.2	22.2	14.5	8.1	4.0	1.5	0.2	0	22.2
Gastrointestinal tract[(b)]	13.6	13.6	13.6	8.9	5.0	2.5	0.9	0.1	0	13.6
Pancreas	3.4	3.4	3.4	2.2	1.3	0.6	0.2	0.0	0	3.4
Breast	25.6	25.6	25.6	16.8	9.4	4.6	1.7	0.3	0	25.6
Bone	6.9	6.7	5.0	2.6	1.6	0.9	0.4	0.1	0	6.9
All other (excluding thyroid)	21.6	19.8	17.1	11.2	6.3	3.1	1.2	0.2	0	21.6
TOTAL (excluding thyroid)	121.6	118.5	105.5	70.1	41.3	22.4	10.0	2.6	0.5	121.6

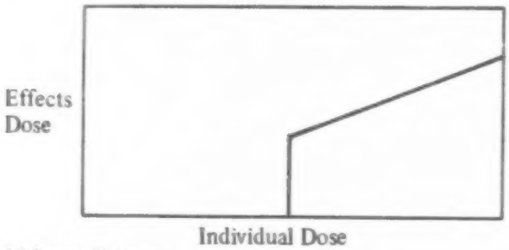
[a] Includes credits for current survival and cure rates.
[b] Includes stomach and rest of alimentary canal.



(a) Upper Estimate



(b) Central Estimate



(c) Lower Estimate

Figure D2. Chronic Health Effects Model for Individuals

Table D15. Dose Effectiveness Factors[15]

Total Dose (rem)	Dose Rate (rem/day)		
	<1	1–10	>10
<10	0.2	(0.2)	(0.2)
10–25	(0.2)	0.4	(0.4)
25–300	(0.2)	(0.4)	1.0

Even though cancer fatalities are calculated on the basis of population dose, applications of the dose thresholds in the lower two types of estimates also require knowledge of the doses received by individuals. In the RSS, the dose to an individual was determined primarily on the basis of an individual's location;* in this study, that dose is essentially independent of location. The doses to individuals from the various liquid pathways depend on the usage habits of the individuals. Those usage habits can vary by orders of magnitude within a population. Furthermore, the populations affected by the various liquid exposure pathways are not necessarily the same. As a result, the doses to populations in terms of their individuals are not easy to describe. Consequently, the linear hypothesis has been used for most calculations in this report. Because the central estimate was most commonly used in the RSS, some

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*In the RSS, an individual's location was determined by the place of residence.

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central estimate calculations have also been performed here. When central estimate results are presented, the alterations which might result from using more realistic usage distributions functions are also indicated.

In this study, appropriate dose effectiveness factors are usually determined separately for each one-year period, on the basis of the total individual dose received during each period. Such a prescription for determining the dose effectiveness factors is probably more realistic than the one utilized in the RSS; however, within the central-estimate framework, it should still overestimate the appropriate factors and hence the resultant health effects. For any atypical release which could result in a large early exposure, the RSS prescription is effectively employed.

D3.3.2.2 Thyroid Effects

In the RSS, the latent and chronic thyroid effects considered were the occurrence of thyroid nodules and the appearance of hypothyroidism. Thryoid nodules can be either benign or malignant. The frequency of occurrence was taken to decrease with increasing dose, with benign nodules being slightly more likely than cancerous ones. In a departure from the RSS, the dose to the thyroid is calculated in this study as the sum of the entire internal dose from all radionuclides except I-131 and the entire external dose. In the RSS, only internal doses during the first 30 days after the accident were included. As in the RSS, only one-tenth of the I-131 dose is counted for determining the incidence of nodules, whereas the entire I-131 dose is counted when considering hypothyroidism. In discussing thyroid neoplasms, a linear no-threshold model has been employed (Table D16); for hypothyroidism, a linear model with a lower dose-threshold has been utilized (Table D16). Due to the relatively short half-lives of most iodine isotopes, the thyroid effects resulting from releases to the liquid pathways are relatively small.

D3.3.2.3 Genetic Effects

Genetic effects result from radiation of the chromosomal matter of the germ cells; that is, radiation increases the rate of mutation in the human population. The effects considered in the RSS included single-gene disorders, multifactorial disorders and chromosomal irregularities. The genetic effects are expressed in the generations after those experiencing the radiation. The amount of genetic damage caused by the radiation depends upon the time elapsed between the exposure and conception. Eventually, unfavorable mutations would probably be deleted from the population. The genetic effects factors developed for the RSS incorporate rates for such elimination and predict the number of affected births for all succeeding generations after the accident. Mutations are counted only if they result in serious disability or spontaneous abortion. The damage is assumed to be entirely from irradiation of the fathers. The health effects factors employed were developed from the BEIR report (Table D17).

Table D16. Expected Total Thyroid Effects per 10^6 Person-rem From Non-I-131 Exposures[a],15

	Dose Range (non-I-131)	Expected Effects per 10^6 Person-rem
Malignant Nodules	≤1500 rem	197.4
	1500 to 5000 rem	98.7
	>5000 rem	0
Benign Nodules	≤1500 rem	134.1
	1500 to 5000 rem	67.0
	>5000 rem	0
Hypothyroidism	<20 rem	0
	>20 rem	110.0

[a] The dose from I-131 is taken to be only one-tenth as effective as the dose from any other radionuclide.

Table D17. Genetic Disorders and Spontaneous Abortions per Rem per 10^6 Persons[a]

	Expected Disorders per Rem per 10^6 Persons[b]						
	From Internal Sources						From External Sources
	Postaccident Period Over Which Dose is Accumulated						
Disorder	0−1	1−10	11−20	21−30	31−40	41−50	
Autosomal Dominant Disorder	45.8	34.5	17.7	4.4	0.7	(c)	45.8
Multifactorial Disorder	85.4	68.6	35.4	8.8	1.3	(c)	85.4
Disorders Due to Chromosomal Aberrations	5.1	4.1	2.1	0.5	0.1	(c)	5.1
Spontaneous Abortions	35.9	28.8	14.9	3.7	0.6	(c)	35.9

[a] The health effects factors given in this table are not quite the same as those listed in the RSS[15] but rather are those listed in the CRAC computer code.

[b] Total is over all time.

[c] Negligibly small in comparison with preceding row.

D3.3.2.4 Time Dependence

Exposure to radiation can be external or internal. For external radiation, removing either the radiation source or the exposed individual terminates the exposure. For internal radiation, the exposure continues regardless of the action; only the eventual elimination of the radiation source from the body terminates the exposure. In general, the calculations of health effects should take this difference in the time dependence of the two types of exposure into account.

In the RSS, internal exposure due to ingestion was calculated as a function of time after consumption. Corresponding health effects factors were obtained as a function of that time. These health effects factors included the change in the age of the exposed population which occurs with the passage of time after ingestion. For example, if ingestion of a radioactive source occurred only immediately after an accident, the population of exposed individuals would gradually be diminished by death. Consequently, the same amount of delivered radiation would become decreasingly effective in causing health effects as time passed.

For all nonthyroid cancers and for all genetic effects, two sets of health effects factors were employed in the RSS, depending upon whether the population could be assumed to have a time-independent age distribution during exposure (last parts of Tables D14 and D17 and Table D16) or whether the population had a time-dependent age distribution during exposure (first parts of Tables D14 and D17). The first type of population is usually appropriate to describe external exposure whereas the second type should be used to consider a singular act of ingestion.

Only one set of health effects factors, that appropriate to describe a time-independent population, was employed for the scoping portions of this study; both time-independent and time-dependent factors were employed in obtaining the final results.

In the RSS, the dose delivered after ingestion was considered as a function of time and was calculated for ten year intervals after the accident, up to 50 years. For most isotopes of importance in this study, the total dose received as a result of ingestion is delivered by the end of the first ten years. (For only one of the dominant isotopes, Sr-90, is the total dose not delivered by then.) In view of this, 50-year dose-commitments and steady-state population health effects factors were employed throughout the scoping portions of this report. Using this combination of dose factors and RSS health effects factors is equivalent to assuming that the entire ingestion dose is delivered at the time of ingestion. The net effect is to slightly increase the calculated health effects with respect to those obtained by the procedure in the RSS.

D3.4 Health Effects Model Uncertainties

The health effects models and the assumptions behind them are discussed in great detail in Chapter 9 and Appendices E through I in Appendix VI of the RSS.15 The primary problem in the health effects models of the RSS can be ascribed to the paucity of appropriate data for assessing the impact of radiological exposure of the human population; that is, adequate data was sometimes not available for deciding between the possible models for certain effects.

Because of the insufficiency of the data, the resulting models have wide ranges of uncertainty. The ranges are broad enough that choices between different possible models often cannot be made. For example, for some types of cancer the data is inadequate to permit a clear choice between a relative-risk cancer model and an absolute-risk one.[21] The difference between these two models can be a factor of several times in overall cancer incidence, with the relative-risk model predicting the higher incidence.

The data is also insufficient to permit a definitive treatment of dose effectiveness factors for low dose rates or low total doses. Opinion varies from believing that low rates or low total doses are less effective in causing cancer to believing just the opposite. This variation could result in at least an order of magnitude difference in the incidence rate calculated for low doses. Many of these variations from the RSS central estimate prescription would result in the prediction of increased incidence rates.

The model used for thyroid effects may not be appropriate.[21] In that model, an interpretation was employed which might tend to underestimate the incidence of thyroid effects. In referenced experiments, the nonappearance of nodules was taken as implying that no effect had occurred, whereas it might have been that total destruction of the tissue had occurred.

As mentioned earlier in this section, the use of constant population health effects factors in conjunction with 50-year dose-commitment factors as was done in the scoping portions of this study will tend to cause slightly higher estimation of health effects than was done in the RSS for internal exposure. This approximation would be responsible for an overestimation of the effects of no more than 50 percent; any overestimation would probably be much less than this.

D3.5 Health Effects Factors and Dose Effectiveness Factors

All of the parameters used in the health effects models were essentially those developed for and used in the RSS.[15] (See Tables D14-17.)

References

[1] U.S. Nuclear Regulatory Commission, "Calculation of Annual Doses to Man from Routine Releases of Reactor Effluents for the Purpose of Evaluating Compliance with 10 CFR Part 50, Appendix I," Regulatory Guide 1.109, Revision 1, USNRC, 1977.

[2] U.S. Nuclear Regulatory Commission, "Liquid Pathway Generic Study," NUREG-0440, USNRC, Appendix C, 1978.

[3] J. F. Fletcher, W. L. Dotson, et al., "HERMES: A Digital Computer Code for Estimating Regional Radiological Effects from the Nuclear Power Industry," HEDL-TME-71-168, Hanford Engineering, 1971.

[4] D. H. Denham and J. K. Soldat, "A Study of Selected Parameters Affecting the Radiation Dose from Radionuclides in Drinking Water Downstream of the Hanford Project," Health Physics 28, 139-144, 1975.

[5] J. K. Soldat, N. M. Robinson and D. A. Baker, "Models and Computer Codes for Evaluating Environmental Radiation Doses," BNWL-1754, BNWL, 1974.

[6] D. K. Todd (Ed.), *Water Encyclopedia*, Water Information Center,Inc., Huntington, NY, 1959.

[7] T. H. Peterson, Jr., "Stable Element and Radionuclide Reconcentration in Aquatic Ecosystems," Fifth Annual Health Physics Midyear Topical Symposium, Idaho Fall, Idaho, 1970.

[8] H. A. Vanderploeg, et al., "Bioaccumulation Factors for Radionuclides in Freshwater Biota," ORNL 5002, ORNL, 1975.

[9] D. E. Reichle, P. B. Dunaway and D. J. Nelson, "Turnover and Concentration of Radionuclides in Food Chains," Nuclear Safety 11, 44-56, 1970.

[10] U.S. Department of Agriculture, *Agricultural Statistics*, USDA, 1974.

[11] D. A. Baker, G. R. Hoenes and J. K. Soldat, "Food - An Interactive Code to Calculate Internal Radiation Doses from Contaminated Food Products," BNWL-SA-5523, BNWL, 1976.

[12] "Handbook of Radiological Protection, Part I: Data,"prepared by a panel of the Radioactivity Advisory Committee (H. J. Dunster, Chairperson), Department of Health and Social Security, Ministry of Health and Social Services Northern Ireland, SNB-11-360079-8, London, England, 1971.

[13] J. K. Nelson, "Distribution of Sediments and Associated Radionuclides in the Columbia River below Hanford," in "Hanford Radiological Sciences Research and Development Annual Report for 1964," D. W. Pearce and J. K. Green, Editors, USAEC Report BNWL-36, BNWL, 1965.

[14] G. L. Toombs and P. B. Cutler (compilers), "Comprehensive Final Report for the Lower Columbia River Environmental Survey in Oregon, June 5, 1961 - July 31, 1967," Oregon State Board of Health, 1968.

[15] U. S. Nuclear Regulatory Commission, *Reactor Safety Study*, WASH-1400, NUREG-75/014, USNRC, Appendix VI, 1975.

[16] National Academy of Sciences, *Radioactivity in the Marine Environment*, NAS, 1971.

[17] G. G. Killough, D. E. Dunning, Jr., and J. C. Pleasant, "INREM II: A Computer Implementation of Recent Models for Estimating the Dose Equivalent to Organs of Man from an Inhaled or Ingested Radionuclide," ORNL/NUREG/TM-84, ORNL, 1978.

[18] G. G. Killough, D. E. Dunning, Jr., S. R. Bernard and J. C. Pleasant, "Estimates of Internal Dose Equivalent to 22 Target Organs for Radionuclides Occurring in Routine Releases from Nuclear Fuel-Cycle Facilities, Vol. I," ORNL/NUREG/TM-190, ORNL, 1978.

[19] G. G. Killough, D. E. Dunning, Jr., S. R. Bernard and J. C. Pleasant, "Estimates of Internal Dose Equivalent to 22 Target Organs for Radionuclides Occurring in Routine Releases from Nuclear Fuel-Sycle Facilities, Vol. II," ORNL/NUREG/TM-XX, ORNL, in review.

[20] K. Eckerman, private communication, ORNL, 1979.

[21] H. W. Lewis, et al., *Risk Assessment Review Group Report to U.S. Nuclear Regulatory Commission*, NUREG/CR-0400, USNRC, 1978.

ADDENDUM D1

Parameters for Pathway Models

Table D1.1. List of Variables for Pathway Models

Symbol	Meaning	Units	Values
A	Fraction of radioactive material with the relatively short weathering constant α	Dimensionless	Table D13
A_i	Amount of isotope i in solution + amount in suspension before treatment	Ci	Calculated
A'_i	Amount of isotope i in solution + amount in suspension after treatment	Ci	Table D4
A_{is}	Amount of isotope i in suspension	Ci	Table D4
A'_{is}	Amount of isotope i in suspension after treatment	Ci	Table D4
A_{iw}	Amount of isotope i in solution	Ci	Table D4
B	Fraction of radioactive material with the relatively long weathering decay constant β	Dimensionless	Table D13
B_{im}	Concentration factor: for aquatic foods, it is the ratio of the concentration of isotope i in organism m to the concentration in the water at equilibrium;	$\frac{(\mu\text{Ci/kg biota})}{(\mu\text{Ci}/\ell\text{ water})}$	Table D6
	for irrigated foods, it is the ratio of the concentration of the isotope i in crop m to the concentration in the soil	$\frac{(\mu\text{Ci/kg wet crop})}{(\mu\text{Ci/kg dry soil})}$	Table D9
$C_{ip}(t)$	Concentration of isotope i in pathway p at time t:		
	$C_{iw}(t)$ is the concentration in untreated water	(μCi/ℓ water)	Calculated
	$C_{id}(t)$ is the concentration in treated water	(μCi/ℓ water)	
	$C_{im}(t)$ is the concentration in the aquatic food m	(μCi/kg food)	
	$C_{im}(t)$ is the concentration in the crop m	(μCi/kg crop)	
	$C_{is}(t)$ is the concentration in the shoreline sediment	(μCi/kg sediment)	
$C^{r'}_{im}(t)$	Related to concentration in soil due to buildup by irrigation	(μCi-yr/ℓ)	Calculated
$C^{s'}_{im}(t)$	Related to concentration on crop m due to buildup to sprinkling	(μCi-yr/ℓ)	Calculated
d_{ip}	Dose factor for isotope i and pathway p:		
	d_{ii} is the ingestion dose factor	rem/μCi	Addendum D3
	d_{is} is the shoreline dose factor	(rem/hr)/(μCi/m^2)	References 1 and 3
	d_{ib} is the swimming dose factor	(rem/hr)/(μCi/ℓ)	References 1 and 3
f	Fraction of year that crop is irrigated	Dimensionless	1.0
f_{ip}	Retention or modification factor for isotope i and pathway p_i:		
	f_{is} = modification factor for portion in suspension for drinking water treatment	Dimensionless	Table D4
	f_{iw} = modification factor for portion in solution for drinking water treatment	Dimensionless	Table D2

Table D1.1. (Continued)

Symbol	Meaning	Units	Values
	f_{im} = retention factor for aquatic food m	Dimensionless	0.31 for vertebrates; 0.45 for crustaceans; and 1.0 for all other invertebrates
	or		
	f_{im} = modification factor for irrigated food m	Dimensionless	Table D11
	f_{is} = shoreline width factor	Dimensionless	Table D12
f^r_{im}	Modification factor for root uptake, that is $(f^s B_{im}/P)$	(kg wet crop/m²)⁻¹	Tables D9 & D11, plus Table E42
f^s_{im}	Modification factor for sprinkling, that is (r/Y_v)	(kg wet crop/m²)⁻¹	Table D11
f_{im}	Modification factor for animal product ingestion, that is, the product of the animal's consumption rate and the transfer coefficient (= ratio of intake rate to concentration in animal product)	Dimensionless	Tables D9 and D10
I	Irrigation rate per area	(ℓ/yr)/m²	Appendix E
K_{di}	Distribution coefficient for isotope i	(Ci/g)/(Ci/ml)	Appendix B
P	"Effective surface density" for soil	kg dry soil/m²	Table D11
$P^k_{pp'}$	Population size of subgroup p′ in pathway p at waterbody k:		
	P^a_p = (population/area) for pathway p at an aquifer	persons/m²	
	$P^l_p(n)$ = population at the nth lake for pathway p	persons	
	P^e_p = (population/length of estuary) for pathway p	persons/m	
	P^o_p = (population/length of ocean shoreline) for pathway p	persons/m	
	$P^r_p(n)$ = (population/length of nth segment of river system) for pathway p	persons/m	
r	Fraction of deposited activity retained on crop	Dimensionless	Table D11
R_f	Fraction of sediment removed by treatment	Dimensionless	Table D2.1
t_p	Time between removal from waterbody and exposure:	yr	
	t_w is the time required for processing and delivery of drinking water		1.0 day
	t_m is the time between harvest of species m and ingestion		Table D11
T	Average lifetime of species under consideration	yr	3
T_m	Translation fraction for crop m	Dimensionless	Table D11
U_p	Usage rate per person for pathway p:		Appendix E
	U_w is the drinking water consumption rate	ℓ/yr	
	U^m_i is the usage rate for crop m	kg/yr	
	U^m_a is the usage rate for aquatic food m	kg/yr	
	U_s is the shoreline usage rate	hr/yr	
	U_b is the swimming rate	hr/yr	
V_f	Volume fraction of sediment	Dimensionless	Appendix B
Y_v	Agricultural productivity	kg wet crop/m²	Table D9
α	Short-time weathering rate	(yr)⁻¹	Table D13
β	Long-time weathering rate	(yr)⁻¹	Table D13

Table D1.1. (Continued)

Symbol	Meaning	Units	Values
λ_{bi}	Biological decay rate for isotope i	$(yr)^{-1}$	Table D7
λ_{ei}	Biological elimination rate for isotope i, that is, $\lambda_{ei} = \lambda_i + \lambda_{bi}$	$(yr)^{-1}$	--
λ_i	Radioactive decay rate ($= .693/\tau_{½\, i}$)	$(yr)^{-1}$	--
λ_r	Uptake rate	$(yr)^{-1}$	.0028 B_{im} (Table D9)
λ_w	Weathering rate	$(yr)^{-1}$	18.07
τ_α	Short-time weathering half-life	yr	Table D13
τ_β	Long-time weathering half-life	yr	Table D13

ADDENDUM D2

Removal of Suspended Sediment by Water Treatment

Many waterbodies tend to have relatively small amounts of suspended sediment. Consequently, the large part of any radioactivity in the water is often in solution. Thus, the removal of the radionuclides from solution is the most important result of water processing in such situations. This removal has been reviewed in Subsection D2.1.2.1. In contrast, river systems often have relatively large amounts of suspended sediment. As a result, a substantial fraction of the radioactivity can reside on the suspended sediment particles. In such cases, the reduction of radioactivity by removal of suspended particles must be considered. This addendum describes a model which addresses such considerations.

The amounts of radioactivity initially in the suspended sediment and in the water of river systems can be written approximately as

$$A_{is} = \frac{A_i K_{di} V_f \rho_s}{1 + K_{di} V_f \rho_s} \quad \text{and}$$

$$A_{iw} = \frac{A_i}{1 + K_{di} V_f \rho_s} , \qquad \text{(D2-1)}$$

respectively, where A_i is the amount of radionuclide i present before treatment, K_{di} is the distribution coefficient for nuclide i (Appendix B, Equation B2-3), V_f is the volume fraction of suspended solids, and ρ_s is the density of the sediment.

Treatment of water from a given river removes a fraction of the suspended sediments, R_f, so that the activity associated with the sediments which remains in the drinking water after treatment is given by

$$A'_{is} = (1 - R_f) A_{is} . \qquad \text{(D2-2)}$$

The sediment removal factors, $\{R_f\}$,are functions of both the original turbidity of the water and the level of water treatment assumed. Using the previous assumptions about the types of water treatment appropriate for the three river systems under consideration in this study (Appendix B, Subsection B3.4), the removal factors for these systems are as follows:

River	Category	Original V_f+	R_f
Tennessee/Ohio	"clear"	1×10^{-5}	0.1*
Lower Mississippi	average	2.3×10^{-4}	0.66
Missouri	turbid	1.6×10^{-3}	0.9

+See Addendum B2 of Appendix B.

*This has been taken to be 0.1 rather than 0.0 (Table D3) so that the effect of a small removal fraction can be investigated; it will be seen that even 0.1 results in a negligible removal of material (Table D2.1).

If the removal from solution is neglected, the total activity remaining after treatment is

$$A'_i = A'_{is} + A_{iw} = \frac{A_i}{1 + K_{di} V_f \rho_s} [1 + (1 - R_f) K_{di} V_f \rho_s] . \qquad \text{(D2-3)}$$

It is implicitly assumed that redistribution of the activity between water and remaining suspended solids takes place only after treatment, that is, that treatment is rapid compared with redistribution. The fraction of activity remaining after filtering can be written as

$$A'_i = A_i \left(1 - \frac{R_f K_{di} V_f \rho_s}{1 + K_{di} V_f \rho_s} \right) . \qquad \text{(D2-4)}$$

The fraction of the initial activity remaining after such sediment removal is shown in Table D2.1 for a range of values of the distribution coefficient for each of the "generic" rivers.

It should be noted that the formula (Equation D2-4) gives a lower fraction of remaining activity (more removal) for a higher initial turbidity. The extremely turbid upper reaches of the Missouri probably require much more extensive filtration than was assumed here. However, the reduction of activity appropriate for the less turbid lower reaches is conservatively assumed for this study.

Table D2.1 Fraction of Initial Radioactivity Remaining After Filtering as a Function of the Distribution Coefficient and the Turbidity

River K_d	Tennessee/ Ohio ($R_f = 0.1$)	Mississippi ($R_f = 0.66$)	Missouri ($R_f = 0.09$)
1	1.00	1.00	1.00
10	1.00	1.00	.99
10^2	1.00	.98	.88
10^3	1.00	.87	.45
10^4	.99	.53	.15
10^5	.95	.35	.11
10^6	.91	.32	.10

The water treatments employed at the different types of sites are summarized in Table D4. In particular, the fractions and the amounts of the initial activity remaining after treatment are given.

ADDENDUM D3

Ingestion Dose Factors

Two different sets of ingestion dose factors have been utilized in the calculations: one generated by INREM II[1-3] and one generated by the dosimetry codes employed in the RSS.[4] The values of the former set of dose factors are listed in References 2 and 3; the values of the latter were generated at ORNL[5] for this study and are listed in this addendum.

The philosophies behind the RSS and the INREM II dosimetry codes are much alike. The INREM II code was developed after the completion of the RSS and was intended to implement the dosimetry models used in the RSS.[5] However, the implementations of the models are not necessarily the same. Also, some of the data employed in the INREM II code is slightly different than those used in the RSS code.[5] In particular, an improved nuclear decay data base plus modified metabolic data bases for some isotopes have been employed in obtaining the INREM II dose factors. The modifications to the metabolic data bases for the INREM II work are thought to be consonant with the overall intent of the RSS work and to represent essentially an updating of the RSS metabolic bases.[6]

Table D3.1. "RSS" Ingestion Dose Factors[a,b]

	Time	Rem/microcurie-ingested					
Nuclide	**Yr**	**T. Body**	**T. Marrow**	**Skeleton**	**Thyroid**	**Lung**	**LLI**
Tc-99	0-10	2.92(-4)[c]	2.89(-4)	2.89(-4)	5.75(-3)	2.89(-4)	1.24(-3)
Ru-103	0-10	5.26(-4)	4.71(-4)	3.23(-4)	1.95(-4)	2.34(-4)	2.26(-2)
Ru-106	0-10	5.98(-3)	4.99(-3)	4.93(-3)	4.87(-3)	4.89(-3)	2.66(-1)
Sb-125	0-10	4.78(-4)	4.36(-4)	3.02(-4)	1.84(-4)	2.17(-4)	2.05(-2)
Te-127M	0-10	7.41(-4)	8.58(-4)	2.21(-3)	2.21(-4)	2.24(-4)	3.87(-2)
Cd-144	0-10	1.01(-3)	1.21(-4)	1.85(-4)	2.71(-6)	6.53(-6)	2.47(-1)
Sm-151	0-10	1.86(-5)	9.11(-7)	2.22(-5)	2.26(-7)	2.27(-7)	3.78(-3)
	0-20	2.09(-5)	1.41(-6)	3.46(-5)	3.51(-7)	3.51(-7)	3.78(-3)
	0-30	2.21(-5)	1.68(-6)	4.14(-5)	4.21(-7)	4.23(-7)	3.78(-3)
	0-40	2.28(-5)	1.84(-6)	4.53(-5)	4.60(-7)	4.62(-7)	3.78(-3)
	0-50	2.32(-5)	1.93(-6)	4.74(-5)	4.82(-7)	4.84(-7)	3.78(-3)
Eu-154	0-10	9.97(-5)	8.56(-4)	6.10(-4)	3.80(-5)	1.79(-4)	6.58(-2)
	0-20	1.03(-4)	8.96(-4)	6.84(-4)	4.41(-5)	1.99(-4)	6.58(-2)
	0-30	1.04(-3)	9.07(-4)	7.04(-4)	4.58(-5)	2.05(-4)	6.58(-2)
	0-40	1.04(-4)	9.10(-4)	7.09(-4)	4.62(-5)	2.06(-4)	6.58(-2)
	0-50	1.04(-4)	9.10(-4)	7.11(-4)	4.63(-5)	2.07(-4)	6.58(-2)
Pu-238	0-10	1.20(-2)	2.36(-4)	1.33(-1)	2.35(-4)	2.35(-4)	1.06(-1)
	0-20	2.17(-2)	4.39(-4)	2.47(-1)	4.37(-4)	4.37(-4)	1.06(-1)
	0-30	3.00(-2)	6.13(-4)	3.46(-1)	6.11(-4)	6.11(-4)	1.06(-1)
	0-40	3.71(-2)	7.64(-4)	4.31(-1)	7.61(-4)	7.61(-4)	1.06(-1)
	0-50	4.31(-2)	8.93(-4)	5.03(-4)	8.90(-4)	8.90(-4)	1.06(-1)

[a] These are ingestion dose factors calculated by the dosimetry codes used for the RSS.

[b] If the dose from the unit intake at time zero had converged within the tabulated integration period, the duplicate entries for longer integration periods were not tabulated.

[c] Read 2.92(-4) as 2.92×10^{-4}.

In general, for a given isotope the dose factors generated with the two different sets of codes and data bases are close, with the exception of the dose factors for α-emitters. The recent upping of the α quality factor from 10 to 20, as recommended by the ICRP, is included in the INREM II work only.[6] Consequently, the INREM II dose factors for α-emitters are approximately two times larger than the corresponding RSS dose factors.

Table D3.2 Location of Metabolic Model Discussions in RSS

Nuclide Outer	Reference Page(a)
Tc-99	D-27
Ru-103	D-28
Ru-106	D-28
Sb-125	D-21
Te-127m	D-28
CE-144	D-22
Sm-151(b)	D-22
Eu-154(b)	D-22
Pu-238	D-20

[a] The page references are for Appendix D of Appendix VI of the RSS.

[b] These elements are grouped with the other rare earths discussed in Section D5 of Appendix D.

References

[1] G. G. Killough, D. E. Dunning, Jr., and J. C. Pleasant, "INREM II: A Computer Implementation of Recent Models for Estimating the Dose Equivalent to Organs of Man from an Inhaled or Ingested Radionuclide," ORNL/NUREG/TM-84, ORNL, 1978.

[2] G. G. Killough, D. E. Dunning, Jr., S. R. Bernard and J. C. Pleasant, "Estimates of Internal Dose Equivalent to 22 Target Organs for Radionuclides Occurring in Routine Releases from Nuclear Fuel-Cycle Facilities, Vol. I," ORNL/NUREG/TM-190, ORNL, 1978.

[3] G. G. Killough, D. E. Dunning, Jr., S. R. Bernard and J. C. Pleasant, "Estimates of Internal Dose Equivalent to 22 Target Organs for Radionuclides Occurring in Routine Releases from Nuclear Fuel-Cycle Facilities, Vol. II," ORNL/NUREG/TM-XX, ORNL, in review.

[4] U. S. Nuclear Regulatory Commission, *Reactor Safety Study*, WASH-1400, NUREG-75/014, USNRC, Appendix VI, 1975.

[5] K. Eckerman, private communication, ORNL, 1979.

[6] G. G. Killough, private communication, ORNL, 1979.

APPENDIX E

Pathway Populations

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APPENDIX E

Pathways and Pathway Populations

Authors: S. J. Niemczyk
W. B. Murfin (Aquatic and irrigated foods)

El Introduction

The Reactor Safety Study (RSS) estimated the risk to the public from potential accidents at 100 nuclear reactors located on 68 U. S. sites.[1] In particular, the potential consequences of releases of radioactivity to the atmosphere were investigated. The 100 reactors considered in the RSS are listed in Table E1, grouped according to the five types of hydrological generic sites addressed in this study.

The exposed populations for the liquid pathways considered in this study are much more difficult to characterize than the populations for the atmospheric pathways were in the RSS. In that study, the atmospheric dose to an individual was assumed to depend only upon the location of the person at the time of the accident. This, in turn, was taken to be determined by the person's place of residence. In contrast, the hydrospheric dose to an individual would be relatively independent of the person's location but instead would depend upon the person's usage habits for all the relevant exposure pathways.

To accurately consider hydrospheric exposure, the distribution functions of the individual usage habits for all significant pathways would be required. No effort has been made to identify appropriate distributions of individual usage rates for any of the pathways. Average usage rates are assumed for all individuals in each pathway.

For each of the five surface-water generic sites, the human populations exposed for each potentially significant pathway and/or the total amount of human exposure for each pathway are estimated in this appendix. In addition, where possible, relevant ranges for each of these populations or amounts of exposure are also indicated.

It is important to note that good population data for the pathways addressed in this study are generally very sparse and difficult to obtain. In some instances, rough approximations have been made to estimate these values. Therefore, the associated uncertainties for many of the values presented in this appendix are quite high, although difficult to quantify. In general, the estimates made for the generic Great Lake Site are probably more reliable than those for the other generic sites. In addition, the population estimates for the internal exposure pathways are usually better than those for the external exposure pathways.

It is impossible to adequately describe a population for a representative site without knowing exactly what the populations are for all the specific sites. The determination of all the specific populations would be a massive undertaking. The "generic" populations used in this study have been chosen by employing the numbers of reactors at various waterbodies as conceptual weighting factors. It is believed that, in general, the populations employed here should be conservative, but not necessarily unrealistically so. Variations of the actual populations for any particular site from the generic ones can be up to several orders of magnitude.

In this study, the following populations have been taken to be representative: Lake Michigan's for a Great Lake; Chesapeake Bay's for an estuary; the Atlantic Coastal Shelf's for an ocean; and both the Tennessee-Ohio-Mississippi and the Missouri-Mississippi Rivers' for river systems. Other populations have also been considered in an attempt to evaluate the "representativeness" of the populations of the chosen waterbodies. Actual dry site* populations have been approximated for all the potentially relevant dry sites and these have been averaged to obtain representative populations.

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*A "dry" site is taken to be one with no relatively close bodies of surface water; that is, it is a site at which contamination of only underground water could probably occur.

Table E1. 100 Reactors (68 Sites) of the Reactor Safety Study[1]

Site	No. of Reactors	Nearest Waterbody	State
1. Great Lakes			
Bailly	1	Lake Michigan	Ind.
Cook	2	Lake Michigan	Mich.
Davis Besse	1	Lake Erie	Ohio
Fermi	1	Lake Erie	Mich.
Fitzpatrick	1	Lake Ontario	N.Y.
Ginna	1	Lake Ontario	N.Y.
Kewaunee	1	Lake Michigan	Wis.
Nine Mile Point	2	Lake Ontario	N.Y.
Palisades	1	Lake Michigan	Mich.
Point Beach	2	Lake Michigan	Wis.
Sterling	1	Lake Ontario	N.Y.
Zion	2	Lake Michigan	Ill.
	16		
2. Estuary			
Calvert Cliffs	2	Chesapeake Bay	Md.
Indian Point	2	Hudson Estuary	N.Y.
Maine Yankee	1	Back River Estuary	Maine
Salem	2	Delaware Estuary	N.J.
	7		
3. Ocean			
Crystal River	1	Gulf of Mexico	Fla
Diablo Canyon	2	Pacific Ocean	Ca.
Forked River	1	Barnegat Bay	N.J.
Millstone	2	Niantic Bay	Conn
Oyster Creek	1	Barnegat Bay	N.J.
Pilgrim	1	Cape Code Bay	Mass.
San Onofre	2	Pacific Ocean	Ca.
Seabrook	1	Atlantic Ocean	N.H.
Shoreham	1	Long Island Sound	N.Y.
St. Lucie	1	Atlantic Ocean	Fla.
Turkey Point	2	Biscayne Bay	Fla.
	15		
4. River System			
Arkansas	2	Arkansas River	Ark.
Beaver Valley	2	Ohio River	Pa.
Browns Ferry	3	Lake Wheeler*	Ala.
Byron	1	Rock River	Ill.
Catawba	1	Lake Whylie*	S.C.
Cooper	1	Missouri River	Neb.
Dresden	2	Illinois River	Ill.
Duane Arnold	1	Cedar River	Iowa
Fort Calhoun	1	Missouri River	Neb.
Grand Gulf	1	Mississippi River	Miss.
Haddam Neck	1	Connecticut River	Conn.
Limerick	1	Schuylkill River	Pa.
McGuire	2	Lake Norman*	N.C.
Monticello	1	Mississippi River	Minn.
North Anna	3	North Anna Reservoir	Va.
Oconee	3	Lake Keowee*	S.C.
Peach Bottom	2	Lake Conowingo*	Pa.
Prarie	2	Mississippi River	Minn.
Quad Cities	2	Mississippi River	Ill.
Riverbend	1	MIssissippi River	La.
Robinson	1	Lake Robinson*	N.C.
Sequoyah	2	Lake Chickamauga*	Tenn.
Summer	1	Lake Monticello*	S.C.
Susquehanna	1	Susquehanna River	Pa.
Three Mile Island	2	Susquehanna River	Pa.
Trojan	1	Columbia River	Ore.
Vermont Yankee	1	Connecticut River	Ver.
Watts Bar	2	Lake Chickamauga*	Tenn.
Zimmer	1	Ohio River	Ohio
	45		
5. Dry			
Belefonte	1	Reservoir on Tenn.	Ala.
Braidwood	1	Kankakee River	Ill.
Brunswick	2	Cape Fear Estuary	N.C.
Farley	2	Chattahoochee River	Ala.
Hatch, E.I.	2	Altamaha River	Ga.
LaSalle	2	Illinois River	Ill.
Midland	1	Tittabawassee River	Mich.
Palo Verde	1	Gila River	Ariz.
Perry	1	Lake Erie	Ohio
Rancho Seco	1	Dry Creek	Ca.
Surry	2	James Estuary	Va.
Waterford	1	Mississippi River	La.
	17		

*Reservoir

The population exposure for any type pathway can be expected to change over an extended period of time for many reasons. Either the size of the population or else the usage habits of its members can vary. This study assumes that for each pathway the product of the population size and the individual usage rate remains constant over the entire time period under consideration. Prediction of exact trends of changes in both populations and usage rates is difficult and is thought to be beyond the scope of this study. However, inasmuch as the population dose for any pathway varies directly with the magnitude of the population usage, the effects of altering the population usage rates can be relatively straightforwardly approximated.

Most pathway usage rates tend to vary approximately in proportion to the size of the general human population. The notable exceptions to this generalization are the rates for the predominately recreational pathways such as those for shoreline usage and immersion in water. The usage rates for these recreational pathways tend to increase much more rapidly than the general population due to trends of both increased numbers of participants and increased usage rates per participant. It is thought that the overall error introduced by the constant population assumption would introduce an error of no more than 25 percent into the results of this study.

The populations presented in this appendix are for the years indicated. No attempt has been made to adjust all the populations to the same year. The error introduced by this procedure is thought to be unimportant relative to that caused by all the other potential sources of error.

This appendix describes the important liquid exposure pathways and the corresponding populations-at-risk. In Section E2, the exposure pathways are discussed qualitatively. In particular, the general factors affecting the resultant doses received by the human population are considered. The potentially-exposed populations for each of the pathways are given in Sections E3 through E7 of this appendix. These populations-at-risk are summarized in the last section of this appendix. (All of the other parameters required for describing the various exposure pathways are presented in Appendix D.)

E2 Pathways

In this study, the five major liquid exposure pathways considered are the following: ingestion of drinking water; ingestion of aquatic foods; ingestion of irrigated foods and related animal products and by-products; shoreline usage; and immersion in water. The most important factors which affect the resultant radiation doses received from each of these pathways can be grouped into two sets: those factors which determine the concentrations of the radionuclides in the portions of the pathways contacted by humans and those factors which determine the human populations-at-risk. The factors dominating the radiation doses are discussed in general terms in this section for the five considered liquid pathways. In particular, those factors affecting the concentrations of radioactivity in the pathways are emphasized. (Those factors determine the populations-at-risk and the populations themselves are described in detail in Sections E3 through E7.)

E2.1 Drinking Water

The drinking water pathway is depicted schematically in Figure E1. The dose received from this pathway is dependent on the concentration of radioactivity in the water. The drinking water pathway includes all the water consumed from drinking either water itself or bottled-water products, and from eating products cooked with water.

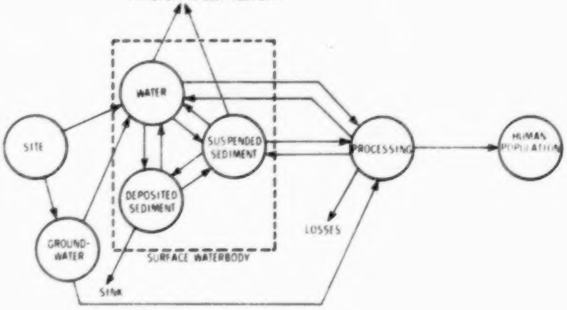

Figure E1. Schematic Diagram of Drinking Water Pathway

Contaminated drinking water can be obtained from either aquifers or surface waterbodies. In the ground, the radioactivity is partitioned between the water and the soil. In general, only the radioactivity in solution in the groundwater is available for drinking, although in certain aquifers some suspended soil particles are present in the water. In any surface waterbody, the radioactivity is divided between the water and the sediment particles. As a result, both the radionuclides in solution and those residing on suspended particles can be ingested.

As was discussed in Chapter 3, the movement of the radionuclides within any waterbody is largely a function of the type of waterbody and its related dilution and flushing characteristics. The drinking water hazard will persist until the radioactivity has been flushed or otherwise removed from the water. In any surface waterbody, even after most of the radioactivity has been removed from the water by flushing, some radioactivity can reenter the water by diffusion from the sediment and/or by elimination from the aquatic foodweb.

After water is taken from the waterbody, it is usually processed by one of several methods. Such processing can remove part or all of the radioactivity from the water. The type of processing employed usually depends both upon the size of the population being served and upon the quality of the water being processed. The activity attached to suspended particles is the most easily removed by processing. The bulk of such activity can usually be removed by simple filtration techniques. At least part of the dissolved radioactivity can often be removed by other standard purification techniques such as coagulation, flocculation, and ion exchange.

Some delay can occur between removal of the water from the waterbody and ingestion. Processing, transport and storage are common sources of such delays. These delays lessen the activity in the water, and therefore the resultant radiation dose, by permitting radioactive decay to occur prior to exposure. For water used directly as drinking water or cooking water, only small delays before usage are generally encountered after removal of the water from the waterbody. In contrast, the delays encountered for bottled products can range from days to months; as a result, significant radioactive decay can occur between bottling and consumption of such products.

The population of water drinkers is distributed approximately in proportion to the national population. Only the water used for bottled products is typically moved significant distances from its source.

E2.2 Aquatic Foods

The aquatic food pathway is shown schematically in Figure E2. The dose received from this pathway is dependent on the concentrations of radioactivity in the aquatic organisms which are consumed by humans. The aquatic food pathway includes human consumption of aquatic foods such as edible portions of finfish, shellfish and other foods obtained from various members of the aquatic foodweb. Such other foods include kelp derivatives and fishmeal. This pathway also includes human consumption of farm and other animals which have been fed with aquatic foods.

The uptake of radioactivity by aquatic organisms is best described in terms of the interactions within the aquatic foodweb. (The term foodweb refers to the complex set of interconnections existing among the various members of an ecosystem.) Uptake of radionuclides by aquatic organisms can be from direct intake of water and/or suspended sediment, from uptake of deposited sediment, from ingestion of other organisms, and from ingestion of detritus. The specific set of interactions undergone and the organisms present at a given site are extremely dependent upon the type of waterbody and its location.

As has been previously mentioned, in approximate steady-state conditons, many radionuclides tend to build up to concentrations in aquatic organisms which are essentially fixed multiples of the concentrations in the water. These "equilibrium" concentrations are assumed to be the maximum possible concentrations which can be present in the organisms for a given steady-state concentration in the water. The rate of buildup in an organism is determined both by the characteristics of the organism and by the specific set of interactions undergone by the organism in the environment.

The buildup of radioactivity in aquatic organisms is dependent upon many factors. Each element has its own distinctive behavior pattern in any given organism. Not only are the various elements sometimes concentrated in different organs or tissues, but they are also often assimilated at different rates. Furthermore, various elements are frequently metabolized differently in many organisms. In addition, the locations of their concentrations can vary from organism to organism. In general, most radionuclides tend to be more concentrated in less highly-evolved species, such as phytoplankton and zooplankton.

Certain aquatic organisms tend to discriminate for some elements, whereas they tend to discriminate against other elements. Such selection occurs throughout the foodweb. As a result of these factors, passage through the foodchain can result in the concentration of significant quantities of certain radionuclides in some aquatic species and the almost complete absence of other radionuclides in edible portions of some members of the aquatic foodweb. Sometimes substantial elimination of radionuclides from the pathway to the human population can also occur either by concentration in an inedible portion of the species, such as the shell, or by elimination from a diet organism in an undigestible form.

Because activity in aquatic species is acquired, in large part, by accumulation up through the foodchain(s), it can take some time before significant

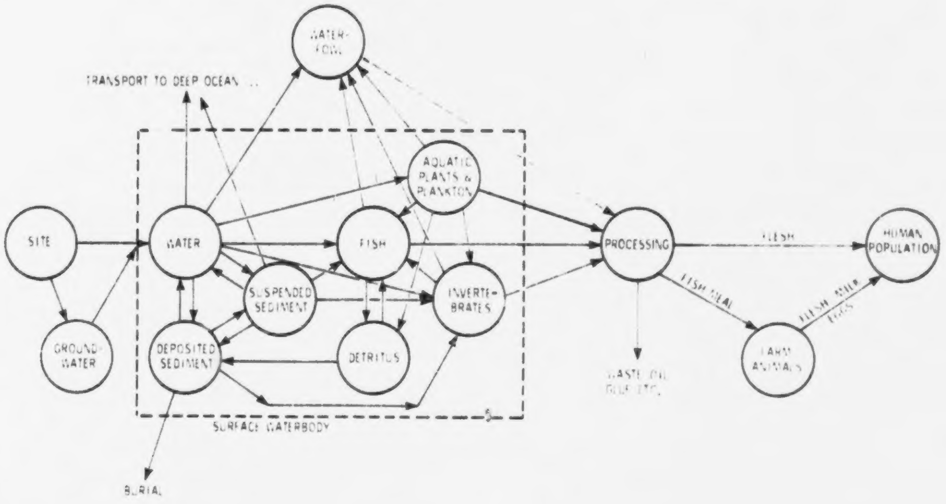

Figure E2. Schematic Diagram of Aquatic Food Pathway

quantities of certain radionuclides build up in certain higher species, such as some finfish. Lower organisms in the foodchain(s), such as some mollusks and crustaceans, tend to build up higher concentrations of radioactivity much more rapidly than some higher organisms. Certain organisms at the bottom of the foodchain(s), such as phytoplankton, tend to come to equilibrium very rapidly, even within hours. The time that it takes the upper members of a foodchain to build up radioactivity is partly a function of the length of the relevant foodchain and partly a function of the types of species which are at the top. It is also a function of the radionuclide under consideration. Even in species at the top of a relatively long foodchain, certain radionuclides can reach their equilibrium concentrations in some higher organisms very rapidly, within days or weeks.

The character of the water is very important in determining the amount of radioactivity which builds up in the aquatic organisms. In general, the radionuclides which would dominate the liquid pathway consequences after a core-melt accident tend to concentrate much more strongly in freshwater organisms than in saltwater ones. As a result, a given level of radioactivity in fresh water usually leads to a higher contamination of the individual aquatic organisms growing there than the same level would in salt water.

Many aquatic species are of a migratory disposition. The regions over which they travel can vary from just local ones to regions of much greater extent. Therefore migrating species can be exposed to drastically different concentrations of radioactivity as they travel in waterbodies which do not have relatively uniform mixing. Concentration gradients due to such lack of mixing can be significant close to the site of any localized release of radioactivity and throughout many waterbodies.

In regions of higher concentrations, the aquatic organisms tend to accumulate radioactivity whereas in regions of lower concentrations, the organisms tend to eliminate some of their radioactive burdens. Thus, even if the concentrations of radioactivity in the water can be described in terms of approximate steady-state behavior, many of the contaminated aquatic organisms might be at some distance from their expected concentration-factor behavior due to their migratory actions

As a result of migration, many organisms which do not reside in the contaminated water can accumulate radionuclides through temporary residence in polluted areas. Therefore, in some waterbodies, the migratory behavior of certain species can cause the contamination of much more of the aquatic ecosystem than would be expected in a cursory consideration.

Because radioactivity can be acquired from ingestion and other uptake of both sediment and detritus, these components of the environment can act as direct secondary sources of radioactivity for the aquatic ecosystem after the bulk of the activity has been flushed from the water in a given region. Such secondary sources are especially important in waterbodies in which the populations of benthic (bottom-dwelling) organisms are significant in the interactions of the aquatic foodweb.

After aquatic foods are harvested, they are usually processed. Such processing generally removes large inedible portions of the organisms, such as shells and bones. As a result, such processing tends to remove those radionuclides which preferentially concentrate in the inedible portions. By the same token it tends to "concentrate" those radionuclides which accumulate in the edible portions. Processing into fishmeal involves the entire organism; thus, no radioactivity is lost by processing into meal.

Some delay, and therefore radioactive decay, can occur between the times of removal of the aquatic organisms from the waterbody and human consumption. For aquatic foods which are eaten fresh, the delays encountered between harvest and ingestion are minimal; at most, they are generally a few days. For foods which are either frozen or canned, the delays before ingestion can be from weeks to months. Dried foods, such as dried fish and kelp derivatives, can also experience significant delays before consumption.

The population of human aquatic food consumers is difficult to locate and describe accurately. Most of the commercial fish catch is probably not eaten close to the source of its harvest; however, the recreational catch tends to be. Variations among individuals tend to make detailed identification of the fish consuming population difficult. Some recreational fishers catch and consume large quantities of certain types of fish from a few selected regions. Other persons of certain ethnic backgrounds tend to eat significant quantities of some specific types of aquatic foods; the sources of the foods can vary greatly. Due to the large size of the commercial catch in some regions, many persons hundreds or even thousands of miles from the scene of the accident might consume small amounts of contaminated aquatic foods.

As a result of these factors, only the total amounts of the various types of aquatic foods which are commercially harvested are readily identifiable; the detailed distribution of that commercial catch is difficult to describe. The recreational catch is more difficult to consider because neither the total amount nor the distribution among individuals is known with any precision.

A significant fraction of the commercial finfish harvest is fed to farm animals. The final human consumers of this part of the fish harvest are obviously even more difficult to describe in detail than those just considered.

E2.3 Irrigated Foods

The irrigation pathway is shown schematically in Figure E3. The dose received from this pathway is dependent on the concentrations of radioactivity in both the irrigated foods and the related animal products and by-products which are consumed by humans. The irrigated food pathway includes all the farm crops grown with irrigation water and all the animal products derived from animals which have been fed either by crops grown with irrigation water or "irrigation" water itself.

The accurate description of the uptake of radioactivity by irrigated plants and the subsequent uptake from plants by animals requires consideration of the complex set of interconnections existing in the agricultural ecosystem. Its complexity is analogous to that found in the aquatic food pathway.

Water for irrigation can be taken from either groundwater or surface-water sources. Both water and suspended sediment can be delivered to the irrigation system. Some water and its radioactive content can be lost from the irrigation system during transport from the waterbody to the fields. However, part of the material lost can ultimately find its way back to either groundwater or surface water.

Irrigation water can contaminate crops in two ways: by deposition onto the plants via sprinkling and by root uptake. Some of the radioactivity sprinkled onto crops is retained while most of the remaining part is weathered off. Some fraction of the radioactive material in the soil can be assimilated by the crops. Part of the remainder is lost from the soil by erosion, leaching, and decay, while part can be taken up by subsequent harvests; another part remains but can be unavailable to the crops because of chemical reactions and/or fixation.

The buildup of radioactivity in irrigated foods is dependent upon many factors. The movement of any radionuclide within the agricultural environment is very element dependent. Certain elements are taken up at different rates by various plants. In addition, the portions of a plant which concentrate certain elements vary drastically from species to species. Some elements are concentrated only slightly, if at all, by certain plants, whereas other elements are concentrated significantly by various plants. The conditions of the soil are extremely important in determining the amount of any element which is incorporated into a plant. The growing conditions are likewise very important.

Cessation of irrigation with contaminated water does not end the irrigation problem. After contaminated water has been used to irrigate a field, the soil is polluted with radioactivity. Even though "clean" water is subsequently used for irrigation, the crops can continue to take up the radionuclides from the soil for many years after the use of polluted water is discontinued.

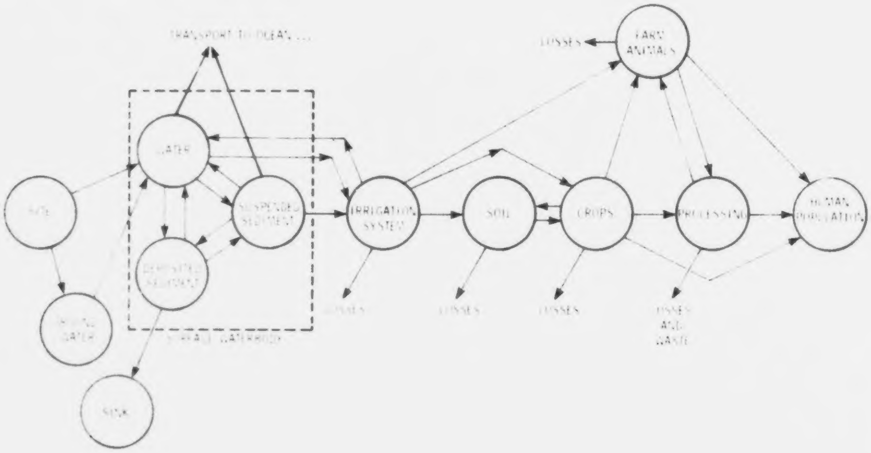

Figure E3. Schematic Diagram of Irrigated Food Pathway

The buildup of radioactivity in animals fed with contaminated crops adds even greater complexity to the overall description of the irrigation pathway. Just as in aquatic organisms and in irrigated plants, the radioactivity can build up for weeks, months, or even years in animals fed with contaminated crops. From such animals the human population obtains both by-products, such as milk and eggs, and direct products, such as flesh. The buildup of certain radionuclides in the by-products can be very rapid. In contrast, the movement of the radionuclides into and between the tissues of the animals commonly utilized for human food is often much slower. Similarly, the elimination of the radionuclides from by-products often occurs relatively rapidly after removal of the radionuclide from the animal's diet. In contrast, the removal of certain radionuclides from the various animal tissues, such as the muscle and bone, frequently occurs only over extended periods of months or years.

Crops intended for human consumption usually undergo some type of processing. The radioactivity deposited onto the crops via sprinkling can be at least partially removed by washing, husking, shelling, and other such preparation techniques. Similarly, the radionuclides which enter the crops through root uptake and concentrate in uneaten portions of the plants or animals can be removed by processing; as a result, crops which are mostly inedible, such as cotton, can have almost all their radioactivity removed by processing. For many of the crops which are fed to animals, processing is rarely used. Consequently, almost all of the radioactivity deposited onto and taken up by such crops can reach the animals.

Various amounts of delay occur between the harvest of the irrigated crops and subsequent consumption by humans of either the crops or the related animal products or by-products. For crops which are eaten fresh by the human population, the delays involved between harvest and ingestion are expected to be on the order of a few days. For crops which are processed before human consumption, the delays can be weeks or months. For crops which are fed to animals, the delays between harvest and consumption of the related animal by-products and products can range from days to years. At the short end of this range are by-products such as fresh milk and eggs; at the long end are direct products such as canned beef and pork.

The population of irrigated food consumers is not easily described without extensive study. Most irrigation in the U.S. is done in relatively less populated regions (Louisiana and west of the Mississippi). However, foods grown with irrigation water are shipped all over the nation. Some people living far from the accident site could receive small amounts of contaminated crops. In contrast, residents living near the sites could receive large amounts of such crops. Only the total amounts of crops which are irrigated are readily identified. A significant fraction of all irrigation water is used for growing food for animals. Most of the irrigated crops fed to animals are probably of rather local origin. However, the animal by-product and product consuming populations are not at all easy to describe in detail.

E2.4 Shoreline Usage

The shoreline pathway is depicted schematically in Figure E4. The dose received from this pathway is dependent on the concentrations of radioactivity in the shoreline and the floodplain sediments. This pathway depends on both the deposition of contaminated suspended sediment onto shoreline and/or

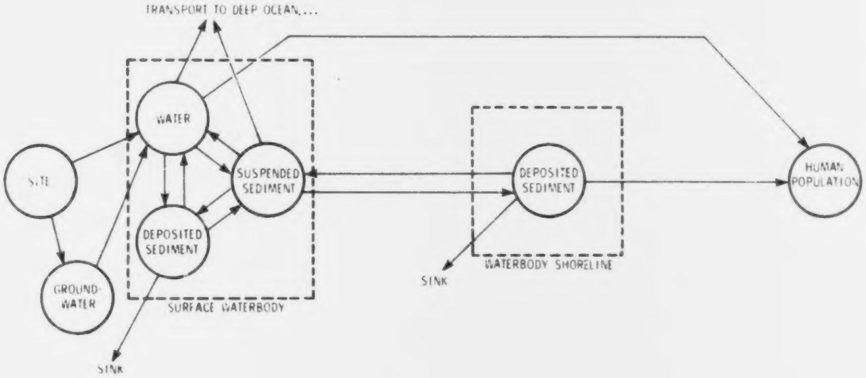

Figure E4. Schematic Diagram of Shoreline and Immersion Pathways

floodplains and the contamination of already-deposited sediment by contact with polluted water. The shoreline pathway encompasses all the external exposure resulting directly from such contamination of shorelines and floodplains. Such exposure includes beach-related portions of swimming, fishing, boating, waterfowl hunting, and walking for pleasure as well as floodplain activities such as farming.

In order for shoreline exposure to be important, contamination of shoreline sediment particles must take place. Flooding and tidal actions by polluted waters are two potential sources of such contamination. Any buildup of contamination is counteracted by various natural "cleaning" processes. For example, flooding, tidal action and various weathering mechanisms, such as rain, can result in the removal of the contaminated sediment and/or its activity. Because most of the buildup and removal processes are extremely time dependent, and often nonuniformly so, the concentration of activity in the sediments can appear to be essentially unrelated to the concentration present in the nearby surface water.

The amount of shoreline exposure depends strongly upon the type of sediment deposited upon the shore. This, in turn, depends upon the type of waterbody. Certain soils and sediments tend to retain and concentrate radioactivity, whereas others do not. Shorelines composed primarily of soils and sediments with significant retention potential can result in much greater shoreline exposure than shorelines with little retention ability.

The character of the water can be very important in determining the amount of radioactivity which builds up in the sediment. In general, the radionuclides which would dominate the liquid pathways after a core-melt accident tend to be adsorbed more strongly to fresh water sediments than to saltwater ones. As a result, a given concentration of radioactivity in fresh water tends to lead to a higher contamination of the associated sediment than the same level in salt water would.

The population of shoreline users is distributed approximately in proportion to the national population. The notable exceptions include populated regions in which there is little surface water available for such exposure and regions in which there are particularly inviting shorelines which might encourage more usage than would otherwise be expected.

E2.5 Immersion

The immersion pathway is depicted schematically in Figure E4. Both the radionuclides in solution and those residing on suspended sediment contribute to the immersion dose. The immersion pathway includes all the external exposure resulting directly from the contamination in the water. Immersion exposure includes swimming and skindiving, along with activities on the surface of the water, such as boating and fishing.

Immersion in artificial pools is often in processed water. Consequently, the exposure received from such swimming can be much less than the exposure obtained from swimming in the surface waterbody from which the pool water is taken.

The population of immersers is distributed approximately in proportion to the national population. The notable exceptions include populated regions in which there is little surface water available for such exposure. However, in such locations, immersion still occurs in artificial pools. Other exceptions are populated areas in which there are especially attractive surface waterbodies. Such areas can experience heavier than normal use.

E3 Drinking Water

The drinking water pathway is taken to include all the water consumed either directly by drinking or else indirectly as a result of using water for cooking and other such purposes. The drinking water contribution to the total population dose depends on both the average per capita consumption rate and the size of the population affected.

The Hermes Study[2] and NRC Regulatory Guide 1.109[3] both recommend the use of average and maximum adult drinking water consumption rates of 1 ℓ/day and 2ℓ/day, respectively. In contrast, the ICRP[4] defined the water intake of a Standard Man as 1.2 ℓ/day of tap water, 1.0 ℓ/day of other fluids such as milk or juice, and 0.3 ℓ/day of water created from the oxidation of foods. The average rate of 1.5 ℓ/day is assumed in this study.

Drinking water populations for the fresh water sites are presented in the succeeding three subsections. Such populations are not presented for the generic estuary and ocean sites; the drinking water populations for these sites are assumed to be zero.

E3.1 Dry Sites

In general, dry sites are not located in urban areas, so that the population which would be affected by groundwater contamination would be rural. Therefore, the average groundwater drinking population density for the vicinity of a dry site is estimated to be equal to the nonurban population density for the

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county in which the site is located. General information on the drinking water sources of the rural populations is not available; however, as a rule, rural areas tend to use groundwater.

From Table E2,[5] it can be seen that the average nonurban population density in the vicinity of dry sites is 31 persons/km^2. The range of average rural population densities for different counties containing the various dry sites is quite large, from 4 to 106 persons/km^2.

Table E2. Population Density at Dry Sites[5] (population density of the county in which the site is located)

Site	Total Population Density (persons/km^2)	Non-urban Population Density (persons/km^2)
Bellefonte[a]	21	13
Braidwood	295	43
Brunswick	28	28
Farley	38	14
Hatch	10	6
La Salle	10	4
Midland	123	46
Palo Verde	106	7
Perry	850	106
Rancho Seco[a]	408	58
Surry	21	21
Waterford[a]	39	20
Average	162	31

[a] This site is near a county line. The populations and areas of both contiguous counties have been used.

These estimates are probably conservative because, in general, the sites listed as "dry" would not be expected to experience any problem with drinking water contamination (see also Subsection E5.1). Most of the sites categorized as "dry" in Table E1 were so-called only because they had a very low average groundwater velocity. In general, these sites are relatively close to the nearest surface waterbody; as a result, no drinking water wells are typically located between the reactor and the nearest surface waterbody at any of these sites.

E3.2 Great Lakes Sites

As is indicated in Table E1, 9 of the non-dry RSS reactors residing next to Great Lakes are located on Lake Michigan, 5 are on Lake Ontario and 2 are on Lake Erie. No reactors are located on Lakes Superior or Huron. For this reason and for certain hydrological reasons,* pathway populations for Lake Michigan are taken as being representative for the generic Great Lakes site. Because Lake Michigan has at least as large a surrounding drinking water population as any of the other lakes, this approach should be conservative.* Estimated populations are also described for each of the other lakes for comparison.

Table E3 presents 1970 U.S. municipal water withdrawal rates and estimated U.S. drinking water populations for each of the Great Lakes. Lake Michigan, which is bordered entirely by the U.S., supplies drinking water for approximately 11×10^6 persons. Each of the other lakes has Canadian as well as U.S. shoreline. As a rough approximation, it is assumed that Lakes Superior and Huron have equal Canadian and U.S. drinking water populations, that Lake Erie has the same total population as Lake Michigan, and that Lake Ontario has 3×10^6 Canadian users (Toronto and surrounding area). As a result, the total drinking water populations are estimated to be as follows: Superior, 0.6×10^6; Huron, 1.4×10^6; Erie, 11×10^6; and Ontario, 5.0×10^6.

In addition to the lake usage, approximately 3.0×10^6 persons live along the St. Lawrence River. All are assumed to use St. Lawrence water as far downstream as Quebec. Below Quebec, the St. Lawrence is estuarial.

Table E4[7] gives the drinking water populations served in each of the Water Resource Regions (WRR's) in the U.S. (Figure E5 shows the WRR's of the contiguous 48 states.) The total number of U.S. users from the Great Lakes (23.4×10^6) is equal to the number given in Table E4 for the Great Lakes WRR. Thus, the numbers assumed in Table E3 correspond to the assumption that 100 percent of the drinking water is taken from surface water. Actually, only approximately 83 percent of the users in that region receive drinking water taken from surface water.

Table E5 lists the nearshore drinking water populations for several unspecified sites on the Great Lakes.[6] The nearshore model used in this study for the Great Lakes extends 25 to 50 km on either side of the site. From the data of Table E5, an average linear population density is estimated to be 1.4×10^3 person/km for a nearshore region 50 km long. Similarly, the average density is approximately 1.9 x 10^3 persons/km for a nearshore region 100 km long. As can be seen by inspection of the data in Table E5, the actual value can vary significantly from this for specific sites.

---

*Of the three Great Lakes with reactors, Lake Michigan has the slowest flushing time. Furthermore, a release into Michigan eventually passes through the succeeding three Great Lakes.

Table E3. Municipal Water Withdrawal from Great Lakes (1970)

Lake	U.S. Withdrawal[a,b] (10^6ℓ/day)	U.S. Population[a,c] (10^6 persons)	Total Population[d] (10^6 persons)
Superior	180	0.3	0.6
Michigan	7700	11.0	11.0
Huron	500	0.7	1.4
Erie	6700	9.4	11.0
Ontario	1400	2.0	5.0

[a] Taken from NRC LPGS.[6]

[b] Great Lakes' Basin Commission, 1975.

[c] Population estimated from withdrawal data assuming per capita usage of 710ℓ/day (190 gallons/day).

[d] Includes estimate for Canadian population.

Table E4. Population Served by Public Water Supplies, 1970

Region Number	Water Resources Council Region	Population Served (10^3persons)	Percentage Served by Surface Water[a]
1	New England	10,100	73
2	Middle Atlantic	33,600	74
3	South Atlantic-Gulf	14,500	45
4	Great Lakes	23,400	83
5	Ohio	15,000	69
6	Tennessee	2,080	75
7	Upper Mississippi	10,900	42
8	Lower Mississippi	4,430	28
9	Souris-Red-Rainy	402	52
10	Missouri	6,710	58
11	Arkansas-White-Red	4,940	64
12	Texas-Gulf	7,920	50
13	Rio Grande	1,370	36
14	Upper Colorado	196	59
15	Lower Colorado	1,790	32
16	Great Basin	1,100	51
17	Columbia-North Pacific	4,630	60
18	California-South Pacific	18,700	57
19	Alaska	126	51
20	Hawaii	694	5
21	Puerto Rico	2,330	83
	United States[b]	165,000	63

Source: Handbook of Water Resources and Pollution Control, Table 2-4.[7]

[a] 100 minus % served by surface water = % served by groundwater.

[b] Including Puerto Rico.

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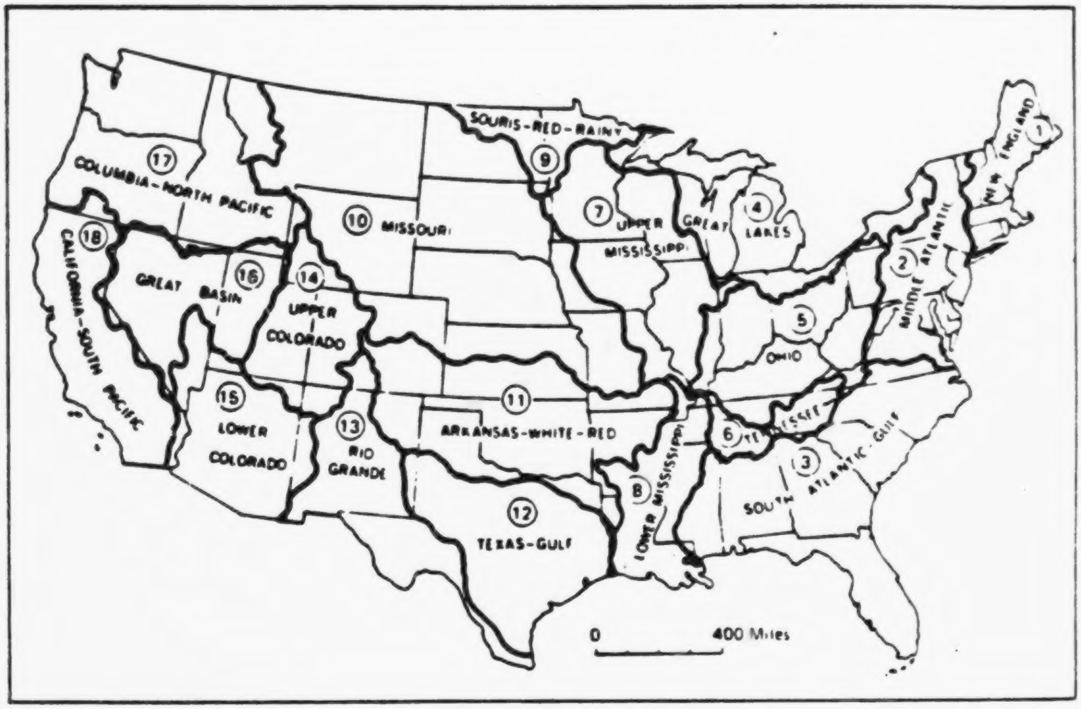
1 NEW ENGLAND
2 MIDDLE ATLANTIC
3 SOUTH ATLANTIC-GULF
4 GREAT LAKES
5 OHIO
6 TENNESSEE
7 UPPER MISSISSIPPI
8 LOWER MISSISSIPPI
9 SOURIS-RED-RAINY
10 MISSOURI
11 ARKANSAS-WHITE-RED
12 TEXAS-GULF
13 RIO GRANDE
14 UPPER COLORADO
15 LOWER COLORADO
16 GREAT BASIN
17 COLUMBIA-NORTH PACIFIC
18 CALIFORNIA-SOUTH PACIFIC
0
400 Miles

Figure E5. Water Resource Regions (WRR's) of the Contiguous 48 States

Table E5. Nearshore Drinking Water Population on Great Lakes[a] (in thousands)

Distance Intervals (km)						
0-8	**8-16**	**16-32**	**32-48**	**48-64**	**64-80**	**Total**
-	-	7.8	21	43	-	72
-	-	-	380	-	-	380
-	14	-	-	-	-	84
79	5	-	-	-	-	84
2	25	-	-	-	-	26
0.75	-	-	-	-	-	0.75
-	25	3	5	-	370	400
-	-	160	-	60	-	220
-	-	140	-	-	52	190
63	190	190	190	3,200	2,700	6,500
Average						
14	26	50	60	330	310	

[a] Based on Table 4.3.2 in the NRC LPGS;[b] the usage in any interval is the total on both sides of the facilities.

E3.3 River System Sites

Table E6 summarizes the drinking water population as a function of downstream distance for 15 unspecified nuclear power plant sites located on river systems.[6] The average values shown for each distance interval include only those sites that lie far enough upstream to have that particular interval. Average linear densities are shown for each site and for each downstream river reach. They range from 100 persons/km to 1,700 persons/km, with an average value of 700 persons/km. These average values were not used in this study; they are included to indicate the approximate range of population data.

Table E7 shows the drinking water population for the two "generic" rivers considered in this study. As might be expected, the population densities are somewhat higher for the two specific rivers than for the averaged densities. The densities for the specific rivers are used for the calculations in this study.

E4 Aquatic Foods

Aquatic foods include the edible portions of finfish and shellfish (crustaceans and mollusks) and food products derived from algae and kelp. They also include some foods obtained from various seagoing mammals and reptiles. (In the data presented in this appendix, the small amounts of whale meat, seal meat, and turtle meat harvested are included with finfish). Both commercial and recreational harvests of aquatic foods for human consumption can be significant.

In addition to the food consumed directly by humans, some aquatic products enter the human food-chain as feed for meat-, milk- and egg-producing farm animals. The aquatic foods consumed by farm animals come primarily from commercial finfish catches.

Table E6. Drinking Water Population on Rivers (in thousands)[a]

	Downstream Distance (km)								Average Linear Density
Site	0-16	16-32	32-80	80-160	160-320	320-640	640-1300	Total	(10^3/km)
1	2.3	0	2.3	1	17	---	---	22	0.1
2	0	15	---	---	---	---	---	15	0.5
3	0	28	0.2	1.5	---	---	---	30	0.2
4	0	0.55	27	8.1	9.2	71	---	120	0.2
5	29	0	8.6	260	36	81	---	410	0.6
6	33	20	51	400	44	---	---	540	1.7
7	0	0	0	0	0	670	26	700	0.5
8	0	300	24	0	280	45	71	720	0.6
9	0	0	7.2	300	240	97	80	730	0.6
10	0	0	6.1	0.4	310	330	85	730	0.6
11	0	0	6.1	7.4	320	310	81	730	0.6
12	0	0	45	0	270	350	380	1000	0.8
13	0	0	0	0	0	330	740	1100	0.8
14	0	0	140	35	100	710	26	1100	0.8
15	0	0	360	0	92	870	31	1400	1.1
Average[b]	4.3	24	48	72	130	350	170	---	---
Average Linear Density (10^3/km)	0.3	1.5	1.0	0.9	0.8	1.1		---	0.7

[a] Taken from the NRC LPGS,[6] Table 4.3.1.

[b] Average includes only those sites that lie far enough upstream to have that particular river reach.

Table E7. Drinking Water Populations for "Generic" Rivers

River	Total Population (millions)	Length (km)	Average Population (persons/km)
Missouri	1.7	1086	1565
Middle Mississippi(a)	1.0	105	9523
Lower Mississippi(b)	1.9	1586	1198
Middle and Lower Mississippi(c)	2.9	1691	1715
Tennessee/Ohio	0.3	849	353
Total	4.9		1351

[a] St. Louis to Cairo

[b] Cairo to New Orleans

[c] St. Louis to New Orleans

In contrast to the case for drinking water, little information is available on the actual individual consumption rates and affected population sizes for the aquatic-food pathway. Instead, the data which is most readily obtainable is for the magnitudes of the aquatic food harvests, that is, for the total population usage rates. It is such data that is considered here.

A large fraction of the commercial finfish catch is devoted to various uses other than direct human consumption; for example, 62% of the 1973 catch was used for such other so-called industrial purposes. The fractions of commercial finfish catches used for industrial purposes for various important fisheries in 1973 are shown in Table E8.[8]

The industrial catch can be processed into pet food, bait, fish meal, fish oils, solubles, and glue. As an example, the breakdown of the disposition of the U. S. finfish catch for industry for one recent year (1973) is presented in Table E9.[8] As can be seen, fish meal constitutes approximately one-fourth of the total industrial catch. It is made from the entire fish and is nearly all fed to livestock.[9] Thus, a substantial fraction of the industrial catch might contribute significantly, although indirectly, to the dose to the human population.

The fraction of mollusks and crustaceans used for industrial purposes is much smaller—2% overall—than the fraction of finfish. Therefore, for the purpose of this study, it will be assumed that all invertebrates are harvested for consumption directly by humans.

Table E8. Finfish Industrial Fractions and Consumption Fractions for Some Important Fisheries

Fishery	Industrial Fraction[8]	Consumption Fraction
New England	.25	.23
Middle Atlantic	.78	.07
Chesapeake	.94	.02
South Atlantic	.59	.13
Gulf	.92	.02
Pacific	.28	.22
Great Lakes	.56	.14
Mississippi River	.12	.27

Table E9. Disposition of the Industrial U.S. Finfish Catch for 1973[8] (live weight in 10^6 kg)

Total Industrial Catch:		
Finfish	1107.9	
Shellfish, etc.	9.2	
Total Catch	1117.1	
Products:		
Pet Food	61.8	6% of catch
Bait	55.4	5% of catch
Meal and Scrap	261.4	23% of catch
Oil	102.1	9% of catch
Solubles and Glue	124.9	11% of catch
Total Products	605.6	54% of catch
Waste	511.5	46% of catch

Due to processing losses, only a fraction of the commercial catch for food for humans is actually consumed. The edible fraction for finfish is defined as the fraction of the commercial finfish catch for food which is actually consumed. From the finfish data presented in Table E10 for the years 1971-1974,[8,10-12] the average edible fraction for all finfish is approximately 0.31. Similarly, from the crustacean data presented in Table E10,[8,10-12] the edible fraction for crustaceans is 0.45. Mollusks are reported as edible meats, so that the edible fraction for mollusks is 1.0.

Table E10. Salient Data for U.S. Commercial Fisheries Catch[8,10-12] (in 10^6 lb)

	Year				
	1971	1972	1973	1974	Average
Total Landings (U.S.)	5018	4806	4858	4967	4912
Finfish for Food	1544		1495	1522	1520
Finfish for Industry	2555	2371	2437	2454	2454
Total Finfish	4099		3932	3976	4002
Crustaceans	721		730	753	735
Mollusks	191		192	232	205
Total Shellfish	913		922	985	940
Algae & Seaweed	3		2	6	4
Pond-raised Fish Meal				13.4	13.4
Exports (Domestic)					
Edible Fish	106.9	107.2	151.0	108.8	118.4
Edible Crustaceans	54.7	53.9	79.4	61.1	62.3
Edible Mollusks	10.2	10.6	8.6	8.2	9.4
Non-edible	229.9	193.2	247.7	199.0	217.4
Imports (for Consumption)					
Non-processed Fish	636.7	945.9	1011.4	943.6	884.4
Processed Fish	826.1	942.7	1022.1	866.7	917.2
Non-processed Crustaceans	184.6	187.2	179.4	173.5	181.2
Processed Crustaceans	81.8	119.9	85.8	107.3	98.7
Mollusks	55.3	71.0	67.1	59.7	63.3
Inedible	623.0	801.7	193.3	296.5	478.6
Seaweed	21.8	34.8	90.8	115.0	65.6

Source: "Fisheries of the United States," various years.[8,10-12]

The consumption fraction (C.F.) for finfish can be defined as the fraction of the total commercial finfish catch which is actually consumed. For any fishery, the consumption fraction is

$$\text{C.F.} = \frac{(\text{edible fraction}) \times (\text{commercial catch for food})}{(\text{total commercial catch})}$$

or,

$$\text{C.F.} = (\text{edible fraction}) \times (1. - \text{industrial fraction}) .$$

The edible fraction includes only losses due to processing, while the consumption fraction includes losses due to both processing and industrial uses.

The finfish consumption fractions for some important fisheries are given in Table E8. Inasmuch as essentially all shellfish are harvested for human consumption, the consumption fractions for crustaceans are taken to be equal to the appropriate edible fraction. Therefore, they are essentially the same for all fisheries (0.45 for crustaceans and 1.00 for mollusks).

The data utilized for estimating the aquatic food harvests are of varying reliabilities. Data for U. S. commercial catches are reasonably reliable for any one year, but may vary considerably from year to year. This is especially true when regional fisheries are considered. For example, harvests of some migratory species such as salmon or menhaden vary by as much as a factor of two from year to year. Data for foreign catches are less reliable than the data for U. S. catches.

The data for recreational catches are extremely unreliable. Because of the difficulty of sampling anglers and the renowned propensity of fishermen to exaggerate the sizes of their catches, the weights of recreational catches must be considered to be only crude estimates. The recreational catch of finfish for food is often greater than the commercial catch; this is particularly true for those areas in which the majority of reactors are located. The unreliability of the data is thus greatest in some of the most important categories and regions.

The harvest densities (i.e., weights of harvest per km or per km²) are even more uncertain than the total weights of harvests. In general, aquatic foods are very heavily harvested from a few relatively small fishing grounds and only sparse catches are taken from other areas.

Because of the great variability in size of catch, industrial fraction, ratio of recreational to commercial catches, and distribution of catches between finfish, mollusks, and crustaceans, it would be misleading to consider just the average of the aquatic food harvest over all fisheries. An estimate of the variation among the catches taken from different fisheries and even among the catches taken from different regions within a given fishery is also important. Therefore values for harvests are presented in this appendix both for averages over large areas (e.g., fisheries and entire waterbodies) and for averages over smaller areas (e.g., portions of fisheries).

The total aquatic food catches for water both surrounding and within the contiguous 48 states and the catches for some closely related waters are summarized in Table E11. The corresponding average aquatic food densities (catches per length or area) are presented in Table E12. Further details about some of these catches and their distributions are presented in the following four subsections and in Tables E13-35.

Table E11. Total (Commercial Plus Recreational) Aquatic Food Catches[a] (live weights of finfish and crustaceans in 10^6 kg/yr, edible meats of mollusks in 10^6 kg/yr)

Fishery	Finfish, Industry	Finfish, Food	Crustaceans	Mollusks
North Atlantic	37	364	30	16
Middle Atlantic	157	168	219	62
South Atlantic	64	219	47	2
Gulf of Mexico	554	311	67	7
California	136	228	4	8
Washington and Oregon	3	116	16	3
Great Lakes	51	70	--[b]	--
Gulf of St. Lawrence	--[b]	230	24	18
Grand Banks	--	202	--	--
Lower Mississippi River[c]	--	4	--	--
Tennessee River[d]	--	14	--	--
Other streams and rivers	--	28[e]	3	--
Lakes, other than Great Lakes	--	284[f]	--	--
Reservoirs and ponds, except commercial ponds and Tennessee River reservoirs	--	270[f]	--	--
Commercial ponds	--	40	--	--
Totals	1002	2548	410	116
Total Commercial Catch	1002	1111	364	108
Total Recreational Catch	0	1437	46	8

[a] Alaska and Hawaii excluded. Data are not all from the same year. Sources of the data are given in the rest of Section E4 and in the corresponding tables, Tables E13 through E35.

[b] Dash indicates data negligible or not available and catch assumed to be negligible.

[c] Excludes tributaries.

[d] Includes reservoirs.

[e] Estimated value, based on extrapolation from 1970 figures (see Subsection E4.4.4).

[f] Estimated value, based on extrapolation from 1970 figures (see Subsection E4.4.4); the total harvest for lakes and reservoirs, excluding the Great Lakes, is estimated to be 568 x 10^6 kg/yr.

Table E12. Summary of Distributed Aquatic Food Data[a]

A. Great Lakes

Lake	Finfish Catches for Food		
	Average (kg/km²-yr)	Nearshore (0 - 8 km) (kg/km²-yr)	Offshore (beyond 8 km) (kg/km²-yr)
Superior	129	249	24
Michigan	285	697	45
Huron	96	239	15
Erie	1340	1340	1340
Ontario	163	310	31

B. Estuaries

Region	Type of Catch			
	Fish, Industry (kg/km²-yr)	Fish, Food (kg/km²-yr)	Crustaceans (km/km²-yr)	Mollusks (kg/km²-yr)
Chesapeake Bay	24600	2800	5010	1450
Delaware Bay	~0	676	1060	128
North Atlantic	6410	25200	2910	1450
Middle Atlantic	6860	3270	2230	1520
South Atlantic	2570	6090	1880	79
Gulf Coast	14030	1720	1200	783
Columbia River				
California Coast	1300	3220	320	1100
Washington and Oregon Coasts	675	9990	1830	660

C. Oceans

C.1 Atlantic and Gulf -

Type	Distance from U. S. Shores		
	0-5 km (kg/km²-yr)	5-20 km (kg/km²-yr)	20-320 km (kg/km²-yr)
Finfish, Industry	2150	350	~0
Finfish, Food	2870	1520	48 (147)[b]
Crustaceans	711	561	15 (19)
Mollusks	116	94	15 (22)

C.2 California -

Type	Distance from U. S. Shores		
	0-5 km (kg/km²-yr)	5-20 km (kg/km²-yr)	20-320 km (kg/km²-yr)
Finfish, Industry	1300	12600	54
Finfish, Food	3220	2510	196
Crustaceans	320	176	2
Mollusks	1100	96	4

C.3 Washington and Oregon -

Type	Distance from U. S. Shores		
	0-5 km (kg/km²-yr)	5-20 km (kg/km²-yr)	20-320 km (kg/km²-yr)
Finfish, Industry	675	0	0
Finfish, Food	9990	1450	85
Crustaceans	1830	550	7
Mollusks	660	0	0

Table E12. (Continued)

C.4 Grand Banks Area -

Type	Distributed Catch (kg/km^2-yr)
Fish, for Food	2250
Crustaceans	28
Mollusks	26

D. Rivers

D.1 Lower Mississippi -

Type	Distribution (kg/km-yr)	Distribution (kg/km^2-yr)
Finfish, Industry	188	215
Finfish, Food	4060	4670
Crustaceans(c)	100	108

D.2 Lower Missouri -

Type	Distribution (kg/km-yr)	Distribution (kg/km^2-yr)
Finfish, Industry	--	--
Finfish, Food	320	428

D.3 Tennessee -

Type	Distrbution (kg/km-yr)	Distrbution (kg/km^2-yr
Finfish, Industry	--	--
Finfish, Food	16000 (total length)	6640 (reservoirs)

D.4 Other Rivers -

Type	Distribution (kg/km-yr)	Distribution (kg/km^2-yr)
Finfish, Industry	--	--
Finfish, Food	230(d,e)	1580(d,f)

E. Miscellaneous

Lakes (other than Great Lakes), reservoirs, and small ponds.	4740 kg/km^2-yr(g)

[a] Details and references are presented in the remainder of Section E4 and in the corresponding tables, Tables E13-E35.

[b] Numbers in parenthesis apply to outer continental shelf north of Cape Hatteras.

[c] Lower 500 km only.

[d] National average (see Subsection E4.4.4).

[e] Assumes the total length of all rivers and streams in the U.S. is 1.4×10^5 km, that is, twice the length of all major rivers in the U. S.

[f] Assumes the total surface area of all rivers and streams in the U.S. is 2×10^4 km^2.

[g] Assumes the total surface area of all small lakes and reservoirs is 12×10^4 km^2.

E4.1 Great Lakes Fisheries

Some general physical data on the Great Lakes are given in Table E13.[13] The American and Canadian commercial catches of finfish from these lakes in 1974 are given in Table E14;[10] the corresponding total finfish catches for food are given in Table E15. The crustacean and the mollusk catches for the Great Lakes are thought to be negligible and so are not considered here.

Substantial quantities of finfish are taken from all of the Great Lakes. The harvests from Lakes Michigan and Erie are particularly large. Except for Lake Michigan, the major portion of the commercial catch from any Great Lake is generally for human consumption. Most of the industrial Lake Michigan catch consists of alewives. The majority of the alewife catch goes into pet foods, and is thus lost (hopefully) from the pathway to the human population. Recreational catches for all the lakes are relatively significant.

The distribution of the catch with respect to distance from shore is not documented. It is assumed for the purpose of estimating the nearshore harvest for Lake Erie that the catch is uniformly distributed over the lake; to obtain that estimate for all other lakes, it is assumed that 90 percent of the catch is taken within 5 miles (9 km) of shore, and the remaining 10 percent is uniformly distributed.

The total amounts of finfish harvested from the Great Lakes and consumed directly by humans are presented in Table E16. These values were obtained from the catches of Table E15 by using the appropriate edible fractions.

Table E13. General Data for Great Lakes[13]

Lake	Volume (km^3)	Surface Area (km^2)	U.S. Shoreline Length (km)	Shoreline Length Total (km)
Superior	12,200	82,100	1,460	4788
Michigan	4,920	57,800	2,250	2668
Huron	3,540	59,600	908	2690
Erie	484	25,700	682	1376
Ontario	1,640	19,000	473	1125

Source: The 1970 World Almanac and Book of Facts.[13]

Table E14. Commercial Catch from Great Lakes for 1974[10]

Lake	Total Commercial Catch		Commercial Industrial Catch		Commercial Food Catch		Commercial Food Catch	
	U.S.	Canada	U.S.	Canada	U.S.	Canada	Total	Distributed
	(10^6 lb)		(10^6 lb)		(10^6 lb)		(10^6 lb)	(kg/km^2)
Superior	5.52	3.51	0	0	5.52	3.51	4.10	50
Michigan	59.60	0	45.5	0	14.10	0	6.41	111
Huron	1.72	3.37	0	0.37	1.72	3.00	2.15	36
Erie	9.83	38.61	0	5.58	9.83	33.03	19.48	758
Ontario	0.32	2.36	0	0.03	0.32	2.33	1.20	63
Totals	76.99	47.85	45.5	5.98	31.49	41.87	33.34	
Totals, U. S./Can. (10^6 lb)	124.85		51.48		73.36			
Totals, U. S./Can. (10^6 kg)	56.75		23.40		33.35			

Table E15. Total Finfish Catches for Food from Great Lakes

Lake	Net Catch: Recreational Catch[a] (10^6 kg)	Net Catch: Commercial Catch (10^6)	Net Catch: Total Catch (10^6 kg)	Distributed Catch: Average (kg/km^2-yr)	Distributed Catch: Nearshore[b] (kg/km^2-yr)	Distributed Catch: Remainder[c] (kg/km^2-yr)
Superior	6.5	4.1	10.6	129	249	24
Michigan	10.1	6.4	16.5	285	697	45
Huron	3.5	2.2	5.7	96	239	15
Erie	15.0	19.5	34.5	1342	1342[d]	1342[d]
Ontario	1.9	1.2	3.1	163	310	31
Total	37.0	33.4	70.4			

[a] 15 x 10^6 kg for Lake Erie, 20 x 10^6 kg for other lakes; catch for other lakes is distributed in proportion to the commercial catch for food.[5]

[b] 90% of total catch divided by nearshore area; nearshore area = (shoreline length) x 8 km.

[c] 10% of total catch divided by (total area - nearshore area).

[d] Lake Erie catch is uniformly distributed.

Table E16. Amount of Great Lakes Finfish Consumed Directly by Humans[a]

Lake	Recreational (10^6 kg/yr)	Commercial (10^6 kg/yr)	Total (10^6 kg/yr)
Superior	2.0	1.3	3.3
Michigan	3.1	2.0	5.1
Huron	1.1	0.7	1.8
Erie	4.7	5.9	10.6
Ontario	0.6	0.4	1.0
Total	11.5	10.3	21.8

[a] Only a negligible amount of the finfish harvest from the Great Lakes is fed to farm animals; consequently, the indirect farm animal pathway is unimportant for the Great Lakes.

E4.2 Estuarine Fisheries

E4.2.1 Atlantic Estuaries

Chesapeake Bay is the most interesting estuary because of both the relatively large number of reactors in the vicinity and the importance of its fishery. Chesapeake Bay is therefore taken to be representative of the "generic" estuary for Atlantic sites. However, the very large size of the total catch and the large fraction of the finfish catch used for industrial purposes are unlike those of any other estuary. Therefore, because of the wide variation among estuaries, the data for Chesapeake Bay should not be blindly utilized to describe all Atlantic estuaries.

The finfish and the shellfish catches for the Chesapeake Fishery for 1973 are shown in Table E17.[11] The recreational catch of finfish was taken to be 1.23 times the commercial food catch.[8] The recreational catch of crustaceans was taken to be 65 percent of the commercial harvest (Tables D-2 and D-3, Reference 14). The recreational take of clams is not precisely known; it was estimated to be 25 percent of the commercial harvest. The recreational harvest of oysters and scallops is believed to be small and was neglected.

The Chesapeake Fishery includes some ocean areas as well as Chesapeake Bay Estuary. The total and the distributed catches for just Chesapeake Bay are also given in Table E17.[11]

For comparison, the Delaware Estuary can also be considered. The entire Delaware catch can be taken to come from Delaware Bay. The total commercial catch for 1973, consisting mostly of edible species, is shown in Table E18.[11] The values for Delaware Bay are believed to be more typical of Atlantic estuaries than those for Chesapeake Bay. Average values for Chesapeake and Delaware Bays together are also presented in Table E18.

The total amounts of seafood harvested from Chesapeake and Delaware Bays and consumed directly by humans and by farm animals are presented in Table E19. The average distributed amounts directly consumed by the human population are also given in that table.

Values for other Atlantic and Gulf estuarine regions can be roughly approximated by the 0-5 km offshore region averages for ocean sites. These values are presented in Subsection E4.3.1.

Table E17. Total Catch for Chesapeake Bay and Chesapeake Fishery (1973)[11]

	Finfish, Industry	Finfish, Food	Crustaceans	Mollusks
Commercial for Fishery (10^6 kg)	239.1	16.4	27.0	36.0
Recreational for Fishery[a] (10^6 kg)	-	20.2	17.6	6.4
Total for Fishery (10^6 kg)	239.1	36.6	44.6	42.4
Total from Bay[b] (10^6 kg)	205.6	23.4	41.9	12.1
Distributed[c] Catch for Bay (kg/km^2)	24593	2800	5013	1445
Farm Animal Feed (kg/km^2)	5632[d]			

[a] Taken to be 1.23 times the commercial food catch; 1.23 represents the ratio of the recreational catch to the commercial food catch for the Middle and North Atlantic.[8]

[b] Taken to be 86% of industrial catch for entire fishery, 65% of edible catch for finfish for fishery, actual quantities of crustaceans and mollusks; recreational catch of crustaceans is 65% of commercial catch.

[c] The area of Chesapeake Bay is taken to be 8360 km^2.

[d] Taken to be 22.9% of the industrial finfish catch based on the nationwide average.

Table E18. Total Catch of 1973 for Delaware Bay[11]

	Commercial (10^6 kg)	Recreational[a] (10^6 kg)	Total (10^6 kg)	Distributed Total Catch[b] (kg/km^2-yr)	Average Distributed Total Catch for Chesapeake and Delaware Bays (kg/km^2-yr)
Finfish, for Food	0.52	0.64	1.16	676	2437[c]
Crustaceans	1.10	0.72	1.82	1060	4388
Mollusks	0.21	0.01	0.22[d]	128	1222

[a] Assumes same ratio for recreational to commercial as Chesapeake Bay.

[b] Area = 1717 km^2 (Reference 15).

[c] Average industrial catch adjusted for abnormally large menhaden catch from Chesapeake Bay ≅5320 kg/km^2 yr.

[d] Surf clams excepted.

Table E19. Amount of Chesapeake Bay and Delaware Bay Seafood Consumed by Humans and Farm Animals

A. Total Amounts Consumed Directly by Humans

	Chesapeake Bay (10⁶ kg/yr)	Delaware Bay (10⁶ kg/yr)
Finfish	7.3	0.4
Crustaceans	18.9	0.8
Mollusks	12.1	0.2

B. Average Distributed Amounts Consumed Directly by Humans

	Chesapeake Bay (kg/km²-yr)	Delaware Bay (kg/km²-yr)
Finfish	868	210
Crustaceans	2256	477
Mollusks	1445	128

C. Total Amounts Consumed by Farm Animals

	Chesapeake Bay (10⁶ kg/yr)	Delaware Bay (10⁶ kg/yr)
Finfish	45.2	0.0

E4.2.2 Pacific Estuaries

The Columbia River Estuary is the most important estuarine fishing area on the Pacific Coast. It is also of interest because of the number of present or planned nuclear reactor sites on the Columbia River. Table E20[10] gives both the total fishing catches for the Columbia River Region and the distributed catches for the Columbia River Estuary for the year 1974.

Analyses of other Pacific estuaries are more difficult owing to the paucity of data on fish catches in such regions. Data are available for catches by states, but cannot easily be particularized to estuarial regions. Values for other Pacific estuarine regions can be very roughly approximated by the 0-5 km offshore region averages for ocean sites. These values are presented in Subsection E4.3.2.

E4.3 Ocean Fisheries

The ocean fisheries include both estuarine regions and nearshore and offshore ocean areas. The catch from these marine fisheries is very dependent upon the distance from the shore. Estimation of the average fish catch for various regions requires that the areas of those regions be approximated. The nearshore and offshore areas used in this report and the assumptions utilized to approximate them are presented in Table E21.[15,16]

Table E20. Catch for 1974 for the Columbia River Region[10,(a)]

A. Total Catch for Region

	Commercial (10⁶ kg/yr)	Recreational (10⁶ kg/yr)	Total (10⁶ kg/yr)
Finfish	23.3(b)	5.1(c)	28.4
Crustaceans	2.5	0.6(d)	3.1
Mollusks	0.0	0.0	0.0

B. Distributed Catch for Estuary(e)

	Commercial (kg/km²-yr)	Recreational (kg/km²-yr)	Total (kg/km²-yr)
Finfish	16640	3640	20280
Crustaceans	1785	430	2215
Mollusks	0	0	0

[a] Washington and Oregon catches are combined.

[b] Essentially all of the finfish were of edible species.

[c] Taken to be 22% of commercial catch.

[d] Taken to be 25% of commercial catch.

[e] Area of maximum harvest is taken to be 700 km²; 50% of the total catch is assumed to come from this area.

Table E21. Ocean Shoreline Lengths, Associated Estuarine Areas, and Approximated Fishing Areas[15,16]

Region	Length of Tidal Shoreline (km)[16,(a)]	Length of General Coastline (km)[16,(b)]	Estuarine Areas (km^2)[15]	Fishing Areas (km^2)		
				0-5 km[(c)]	5-20 km[(d)]	20-320 km[(e)]
North Atlantic	9,860	761	1,558	5,363	11,415	228,300
Middle Atlantic	17,553	689	18,725	22,169	10,335	206,700
South Atlantic	19,187	1,880	13,480	22,880	28,200	564,000
Gulf of Mexico	27,580	2,625	25,074	38,199	39,375	309,300
Pacific[(f)]	64,760	12,250	47,850	109,100	183,750	3,675,000
California	3,427	523	2,233	4,848	7,845	156,900
Oregon	1,410	476	233	2,613	7,140	142,800
Washington	3,026	252	784	2,044	3,780	75,600
Total[(g)]	82,043	7,206	62,087	98,116	108,090	1,683,600

[a] The tidal shoreline includes bays, sounds, rivers, and creeks to a point where the tidal water narrows to 100 feet.

[b] The general coastline follows the apparent outline of the coast; the coastlines of sounds and bays is included only to a point where they narrow to 30 nautical miles.

[c] Length of general coastline times 5 km, plus associated estuarine areas.

[d] Length of general coastline times 15 km.

[e] Length of general coastline times 300 km.

[f] Includes Alaska.

[g] Excludes Alaska.

E4.3.1 Atlantic Ocean and Gulf of Mexico

Table E22[11] summarizes both the commercial and the total Atlantic and Gulf of Mexico catches as functions of the distance caught offshore. As expected, the catches are very dependent upon that distance.

Much of the aquatic-food harvest data for the Atlantic Coast is given in terms of the harvests for three main regions: the North Atlantic, the Middle Atlantic and the South Atlantic. The North Atlantic region includes the waters from Maine through Connecticut; the Middle Atlantic region extends from New York through Virginia; and the South Atlantic region includes from North Carolina through the Florida East Coast. The commercial, recreational, total and distributed total catches for the North Atlantic, the Middle Atlantic, the South Atlantic and the Gulf of Mexico are given in Tables E23-E27.[8,17]

The estimated total recreational catches of finfish for the North Atlantic and Middle Atlantic regions are 123 percent of the corresponding U. S. commercial catches of finfish for food.[8] These recreational catches have been distributed according to the following formulae:

$$\text{(recreational catch, 0-5 km)} = \frac{\begin{pmatrix}\text{total recrea-}\\ \text{tional catch}\end{pmatrix} \times \begin{pmatrix}\text{commercial}\\ \text{catch, 0-5 km}\end{pmatrix}}{\begin{pmatrix}\text{commercial}\\ \text{catch, 0-5 km}\end{pmatrix} + \begin{pmatrix}\text{commercial}\\ \text{catch, 5-20 km}\end{pmatrix}} ;$$

$$\text{(recreational catch, 5-20 km)} = \frac{\begin{pmatrix}\text{Total recrea-}\\ \text{tional catch}\end{pmatrix} \times \begin{pmatrix}\text{commercial}\\ \text{catch, 5-20 km}\end{pmatrix}}{\begin{pmatrix}\text{commercial}\\ \text{catch, 0-5 km}\end{pmatrix} + \begin{pmatrix}\text{commercial}\\ \text{catch, 5-20 km}\end{pmatrix}} .$$

The estimated recreational catch of finfish for the South Atlantic region is 520 percent of the U. S. commercial catch of finfish for food;[8] the estimated recreational catch of finfish for the Gulf of Mexico is 260 percent of the U. S. commercial catch of finfish for food.[8] The recreational catches for the South Atlantic and the Gulf of Mexico have been distributed in proportion to the commercial catches. The distribution of catches with respect to distance from shore has been handled differently in the South Atlantic and Gulf regions than in the North and Middle Atlantic because there are many recreational charter fishing firms in the southern regions that take fishing parties to considerable distances off shore.

For the entire Atlantic and for the Gulf of Mexico, the recreational catch of crustaceans is 65 percent of the commercial catch of crabs in the area from 0-5 km. It is believed that little or no recreational catch of crustaceans occurs at distances greater than 5 km from the coast. The recreational catch of mollusks is taken to be 25 percent of the commercial harvest of clams from the 0 to 5 km area. The U. S. commercial harvests of both finfish and shellfish from the Atlantic and the Gulf of Mexico are presented in Table E23; the corresponding recreational harvests are summarized in Table E24.

The foreign catch of finfish in the North Atlantic fisheries[17] is 210 percent of the U. S. catch in that region. In the same area, the foreign catch of crustaceans is 26 percent of the U. S. catch, and the foreign catch of mollusks is 49 percent of the U. S. catch. The North Atlantic foreign percentages have been used for both the North Atlantic and the Middle Atlantic fisheries. The foreign catches for both finfish and shellfish have been assumed to be negligible for both the South Atlantic and Gulf of Mexico fisheries. The Atlantic and the Gulf of Mexico foreign harvests are summarized in Table E25.

The total Atlantic and Gulf harvests for the four main fisheries are summarized in Table E26. The corresponding values for the distributed harvests are presented in Table E27. In general, the Atlantic and the Gulf of Mexico nearshore catches include substantial contributions from estuarine regions. Consequently, the harvests presented in this subsection tend to greatly overestimate the catches from nonestuarine nearshore ocean regions.

The total amounts of seafood ingested directly by the human population from Atlantic and Gulf Fisheries are presented in Table E28. These amounts were obtained by combining the catches given in Tables E22-E27 with appropriate edible fractions. The amounts of finfish ingested by farm animals are also given in Table E28.

Table E22. Total (1974) Atlantic and Gulf Catches

A. Commercial U.S. Landings

	Distance Off U.S. Shores			
	0-5 km (10^6 kg)	5-20 km (10^6 kg)	20-320 km (10^6 kg)	Total(a) (10^6 kg)
Finfish, Industry(b)	781.2	31.2	0.0	869.9
Finfish, Food	144.8	57.2	85.7	466.9
Crustaceans	109.8	24.0	16.4	160.2
Mollusks	43.4	8.4	18.7	70.5

[a] Columns will not add to total. Difference is due to catch at greater than 320 miles and catch taken from foreign shores.

[b] All alewives and menhaden.

B. Total (Commercial and Recreational) U.S. Landings

	Distance Off U.S. Shores			
	0-5 km (10^6 kg)	5-20 km (10^6 kg)	20-320 km (10^6 kg)	Total (10^6 kg)
Finfish, Industry	781.2	31.2	0.0	869.9
Finfish, Food(a)	342.6	135.4	202.8	1104.8
Crustaceans(b)	181.2	50.0	26.4	267.6
Mollusks(c)	42.5	8.4	27.3	200.0

[a] Recreational catch = 637.9 x 10^6 kg.[10] Distributed in same ratio as commercial catch.

[b] Recreational catch = 65% of nearshore commercial catch.

[c] Recreational catch = 25% of nearshore catch of clams.

Table E23. U.S. Atlantic and Gulf Commercial Catches (Estuarine Plus Ocean)[a] (in 10^6 kg)

	North Atlantic (Maine-Conn, inclusive)			Middle Atlantic (NY-Va, inclusive)			South Atlantic (NC-Fla. East Coast, inclusive)			Gulf of Mexico		
	0-5km	5-20km	20-320km	0-5km	5-20km	20-320km	0-5km	5-20km	20-320km	0-5km	5-20km	20-320km
Fish, Industry	34.4	2.9	0	152.1	5.2	0	58.7	5.0	0	536.0	18.1	0
Fish, Food	41.1	25.5	57.5	28.8	4.3	7.6	21.1	7.7	6.6	53.8	19.7	14.0
Crustaceans	12.0	7.6	5.0	30.0	2.4	0.8	26.1	3.8	0	41.7	10.2	10.6
Mollusks	6.0	1.7	4.0	28.8	6.2	14.7	1.7	0.1	0	6.9	0.4	0

Table E24. Atlantic and Gulf Recreational Catches (Estuarine Plus Ocean)[a] (in 10^6 kg)

	North Atlantic (Maine-Conn, inclusive)			Middle Atlantic (NY-Va, inclusive)			South Atlantic (NC-Fla. East Coast, inclusive)			Gulf of Mexico		
	0-5km	5-20km	20-320km	0-5km	5-20km	20-320km	0-5km	5-20km	20-320km	0-5km	5-20km	20-320km
Fish, Industry	0	0	0	0	0	0	0	0	0	0	0	0
Fish, Food	75.0	46.6	0	97.4	14.5	0	109.4	39.9	34.2	139.8	49.7	35.3
Crustaceans	3.6	0.5	0	19.5	0	0	17.0	0	0	4.2	0	0
Mollusks	1.8	0	0	4.9	0	0	0.1	0	0	0.1	0	0

Table E25. Atlantic and Gulf Foreign Commercial Catches (Estuarine Plus Ocean)[17,(a),(b)] (in 10^6 kg)

	North Atlantic (Maine-Conn, inclusive)			Middle Atlantic (NY-Va, inclusive)			South Atlantic (NC-Fla. East Coast, inclusive)			Gulf of Mexico		
	0-5km	5-20km	20-320km	0-5km	5-20km	20-320km	0-5km	5-20km	20-320km	0-5km	5-20km	20-320km
Fish, Industry	0	0	0	0	0	0	0	0	0	0	0	0
Fish, Food	0	0	118.4	0	0	15.7	0	0	0	0	0	0
Crustaceans	0	0	1.3	0	0	0.2	0	0	0	0	0	0
Mollusks	0	0	2.0	0	0	7.2	0	0	0	0	0	0

[a] Taken from International Commission for the North Atlantic Fisheries (ICNAF) subareas 4X, 5Y, 5Z, 6A, 6B, and 6C[17] (Figure E6); these ICNAF areas do not precisely follow any particular offshore distance; however, the majority of the catch comes from the continental shelf.

[b] There is no foreign fishing very close to US coasts; the entire foreign catch has been assigned to the area from 20 to 320 km offshore.

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Table E26. Total Atlantic and Gulf Catches (Estuarine Plus Ocean)[8,17,(a)] (in 10^6 kg)

	North Atlantic (Maine-Conn, inclusive)			Middle Atlantic (NY-Va, inclusive)			South Atlantic (NC-Fla. East Coast, inclusive)			Gulf of Mexico		
	0-5km	5-20km	20-320km	0-5km	5-20km	20-320km	0-5km	5-20km	20-320km	0-5km	5-20km	20-320km
Fish, Industry	34.4	2.9	0	152.1	5.2	0	58.7	5.0	0	536.0	18.1	0
Fish, Food	116.1	72.1	175.9	126.2	18.8	23.3	130.5	47.6	40.8	193.6	67.8	49.3
Crustaceans	15.6	8.1	6.3	49.5	2.4	1.0	43.1	3.8	0	45.9	10.2	10.6
Mollusks	7.8	1.7	6.0	33.7	6.2	21.9	1.8	0.1	0	7.0	0.4	0

[a] These values do not correspond to those in Table E22 because they are from different sources and because they are for different years.

Table E27. Distributed Total Atlantic and Gulf Catches (Estuarine Plus Ocean) (in kg/km²-yr)

	North Atlantic (Maine-Conn, inclusive)			Middle Atlantic (NY-Va, inclusive)			South Atlantic (NC-Fla. East Coast, inclusive)			Gulf of Mexico		
	0-5km	5-20km	20-320km	0-5km	5-20km	20-320km	0-5km	5-20km	20-320km	0-5km	5-20km	20-320km
Fish, industry	6414	254	0	6869	503	0	2566	177	0	14032	460	0
Fish, Food	21648	6316	770	5693	1819	113	5704	1688	72	5068	1722	159
Crustaceans	2909	710	28	2233	232	5	1884	135	0	1202	259	34
Mollusks	1454	149	26	1520	600	106	79	4	0	183	10	0

Table E28. Amount of Atlantic and Gulf of Mexico Seafood Consumed by Human and Farm Animals

	Amounts Consumed by Humans (in 10^6 kg)			Amounts Consumed by Farm Animals[a] (in 10^6 kg)		
A. North Atlantic	Distance Off U.S. Shores			Distance Off U.S. Shores		
	0-5 km	5-20 km	20-320 km	0-5 km	5-20 km	20-320 km
Finfish	36.0	22.4	54.5	7.9	0.7	0.0
Crustaceans	7.0	3.6	2.8	---[b]	---	---
Mollusks	7.8	1.7	6.0	---	---	---
B. Middle Atlantic	Distance Off U.S. Shores			Distance Off U.S. Shores		
	0-5 km	5-20 km	20-320 km	0-5 km	5-20 km	20-320 km
Finfish	39.1	5.8	7.2	35.5	1.2	0.0
Crustaceans	22.3	1.1	0.5	---	---	---
Mollusks	33.7	6.2	21.9	---	---	---
C. South Atlantic	Distance Off U.S. Shores			Distance Off U.S. Shores		
	0-5 km	5-20 km	20-320 km	0-5 km	5-20 km	20-320 km
Finfish	40.5	14.8	12.6	13.4	1.1	0.0
Crustaceans	19.4	1.7	0.0	---	---	---
Mollusks	1.8	0.1	0.0	---	---	---
D. Gulf of Mexico	Distance Off U.S. Shores			Distance Off U.S. Shores		
	0-5 km	5-20 km	20-320 km	0-5 km	5-20 km	20-320 km
Finfish	60.0	21.5	15.3	122.7	4.1	0.0
Crustaceans	20.7	4.6	4.8	---	---	---
Mollusks	7.0	0.4	0.0	---	---	---

[a] Taken to be 22.9% of the industrial finfish catch, based on the nationwide average.

[b] Dashes denote a lack of data which has been taken to indicate a negligible contribution.

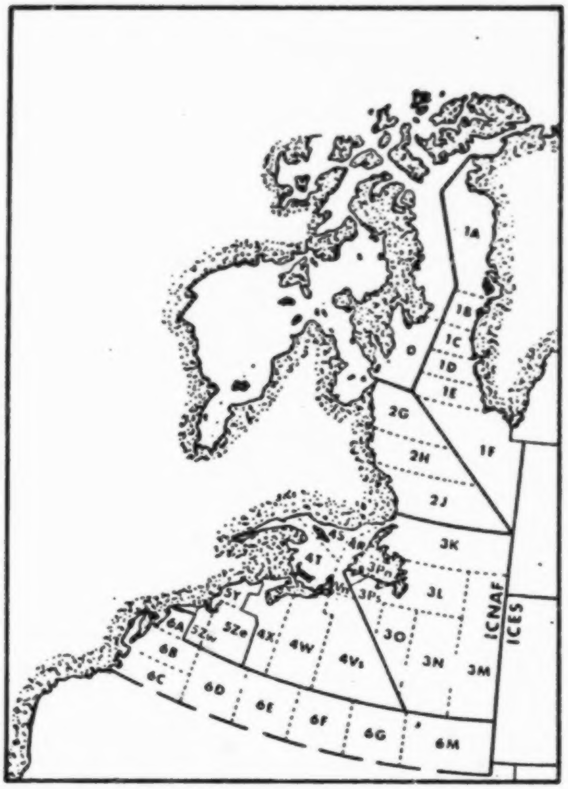

Figure E6. ICNAF Subareas for the North Atlantic

E4.3.2 Pacific Ocean

Fisheries of the Pacific Coast are neither as clear-cut nor as well documented as those of the Atlantic and the Gulf coasts. A major difference is that almost all Pacific catches (except for anchovies and herring) are destined for human food. A relatively small amount of scrap is used for animal food or fish meal.

Data are not available on the separate landings from California, Oregon, and Washington as functions of distance offshore; however, data are available concerning the species composing the catch for each state. Consequently, an estimate of the harvests as functions of distance caught from shore can be made by assuming that the catches from these states are equal to the total Pacific catch as a function of distance, times the fraction (for each species) of the catches from these states. This information is summarized in Tables E29[11] and E30.[11] California has been separated from Washington and Oregon in these tables because of the very different nature of the catches in these two regions.

For California, the recreational catch is taken to be the following: 22 percent of the commercial catch of finfish,[8] distributed in the same proportion by distance as the commercial catch; 25 percent of the commercial clam harvest, as was estimated for Atlantic estuarine fisheries; and negligible for crustaceans. For Washington and Oregon, the recreational finfish catch is taken to be 43 percent of the commercial catch of finfish;[8] the recreational crustacean and mollusk catches are assumed to be negligible.

The total and the distributed (commercial plus recreational) catches for California are given in Table E31;[11,18] the total and distributed Washington-Oregon catches are given in Table E32.[11] The amounts of seafood ingested directly by humans and by farm animals are presented in Table E33.

There are few, if any, important estuarine fisheries in California. Thus, the values for the California nearshore catch are fairly representative of those for nonestuarine regions. In contrast, much of the Washington and Oregon nearshore salmon catch is undoubtedly from estuaries. Therefore, a significant fraction of the nearshore ocean catch from these two states is actually estuarine in origin.

E4.3.3 Gulf of St. Lawrence

Because all of the water from the Great Lakes eventually empties into the Gulf of St. Lawrence, the aquatic food catch from that gulf might be important. Total catches from the Gulf in 1974 are given in Table E34.[17,19] Only small quantities of industrial fish are taken from this region; essentially the entire catch is intended for human consumption. Gulf of St. Lawrence water could eventually affect the important fisheries on the Grand Banks. The total and the distributed catches of finfish from the banks are also given in Table E34.

E4.4 River System Fisheries

The most important commercial riverine fishery is the Mississippi River and its tributaries. Recreational fisheries are generally much more important than commercial fisheries on river systems.

Table E29. Pacific Ocean Fishery Catch for 1973[11]

Species	Pacific Catch (10^3 lb)	California Catch (10^3 lb)	Washington and Oregon Catch (10^3 lb)	Fraction from California	Fraction from Washington and Oregon
Anchovies[a]	277,712	277,704	8	1.00	0.00
Bonito	30,788	30,788	-	1.00	0.00
Cod	8,419	-	8,261	0.00	.98
Flounder	54,505	30,397	3,445	.56	.43
Halibut	26,322	-	1,571	0.00	.06
Hake	2,594	1	2,560	0.00	1.00
Herring[a]	44,592	2,820	6,901	.06	.15
Mackerel	20,736	20,736	-	1.00	0.00
Lingcod	8,378	4,638	3,559	.55	.42
Ocean Perch	6,428	-	6,428	0.00	1.00
Rock Fish	40,202	22,216	17,797	.55	.44
Sablefish	12,366	8,550	1,905	.69	.15
Salmon	221,646	9,669	75,484	.04	.34
Skipjack	1,392	1,392	-	1.00	0.00
Tuna	331,240	292,336	38,904	.88	.12
Other Fish	10,449	6,650	1,750	.64	.17
Dungeness Crab	14,373	1,023	6,947	.07	.48
King Crab	76,824	-	-	0.00	0.00
Snow Crab	61,719	-	-	0.00	0.00
Shrimp	152,226	2,762	29,931	.02	.02
Other Crustaceans	1,705	1,302	16	.76	.01
Clams	1,212	374	741	.31	.61
Oysters	6,599	727	5,872	.11	.89
Squid	12,129	12,129	-	1.00	0.00
Other Mollusks	4,394	4,305	89	.98	.20

[a] Largely used for industrial purposes.

Table E30. Pacific Commercial Catches for 1973 (Estuarine Plus Ocean)[11] (in 10^6 kg)

A. California

	Distance Off U.S. Shores			
Type	**0-5 km**	**5-20 km**	**20-320 km**	**Total**
Finfish, Industry	6.3	99.1	8.5	135.9
Finfish, Food	12.7	16.0	24.9	185.0
Crustaceans	1.54	1.38	.33	3.8
Mollusks	5.30	.75	.68	7.68
Seaweed (Kelp)	---	---	---	108.0[19,a]

B. Washington and Oregon

	Distance Off U.S. Shores			
Type	**0-5 km**	**5-20 km**	**20-320 km**	**Total**
Finfish, Industry	3.14	0.0	0.0	3.14
Finfish, Food	32.6	11.1	13.0	84.3
Crustaceans	8.5	6.0	1.55	14.8
Mollusks	3.0	0.0	0.0	3.03

[a] The distribution of kelp with respect to distance offshore was not given; most of the kelp is harvested relatively close to shore; this harvest was taken almost entirely between San Diego and Los Angeles; the enormous Mexican kelp harvest just south of San Diego is not included.[18]

Table E31. Total and Distributed 1973 Commercial Plus Recreational California Catches (Estuarine plus Ocean)[11]

A. Total (Commercial plus Recreational) Catches (in 10^6 kg/yr)

	Distance Off U.S. Shores		
Type	0-5 km	5-20 km	20-320 km
Finfish, Industry	6.3	99.1	8.5
Finfish, Food(a)	15.6	19.7	30.7
Crustaceans	1.54	1.38	0.33
Mollusks	5.34	0.75	0.68

B. Distributed (Commercial plus Recreational) Catches (in kg/km^2-yr)

	Distance Off U.S. Shores		
Type	0-5 km	5-20 km	20-320 km
Finfish, Industry	1300	12632	54
Finfish, Food	3218	2511	196
Crustaceans	320	176	2
Mollusks	1100	96	4

[a] Recreational catch = 42.8×10^6 kg for the year given in Reference 11; distributed in the same ratio as the commercial catch for food for 1973.

Table E32. Total and Distributed 1973 Commercial plus Recreational Washington and Oregon Catches (Estuarine Plus Ocean)[11]

A. Total (Commercial plus Recreational) Catches (in 10^6 kg/yr)

	Distance Off U.S. Shores		
Type	0-5 km	5-20 km	20-320 km
Finfish, Industry	3.14	0.0	0.0
Finfish, Food(a)	46.5	15.8	18.6
Crustaceans	8.5	6.0	1.55
Mollusks	3.08	0.0	0.0

B. Distributed (Commercial plus Recreational) Catches (in kg/km^2-yr)

	Distance Off U.S. Shores		
Type	0-5 km	5-20 km	20-320 km
Finfish, Industry	675	0	0
Finfish, Food	9985	1447	85
Crustaceans	1830	550	7
Mollusks	660	0	0

[a] Recreational catch = 36×10^6 kg, including Alaska, the number of saltwater anglers fishing Alaskan ocean waters is insignificantly small,[11] so the entire catch is assigned to Washington and Oregon; the recreational catch is distributed in the same ratio as the commercial catch for food.

Table E33. Amount of Pacific Seafood Consumed by Humans and Farm Animals

A. California

	Amounts Consumed by Humans (in 10^6 kg)			Amounts Consumed by Farm Animals(a) (in 10^6 kg)		
	Distance Off U.S. Shores			Distance Off U.S. Shores		
	0-5 km	5-20 km	20-320 km	0-5 km	5-20 km	20-320 km
Finfish	4.8	6.1	9.5	1.4	22.7	1.9
Crustaceans	0.7	0.6	0.1	---	---	---
Mollusks	5.3	0.8	0.7	---	---	---

B. Washington and Oregon

	Distance Off U.S. Shores			Distance Off U.S. Shores		
	0-5 km	5-20	20-320	0-5 km	5-20 km	20-320 km
Finfish	14.4	4.9	5.8	0.7	0.0	0.0
Crustaceans	3.8	2.7	0.7	---	---	---
Mollusks	3.1	0.0	0.0	---	---	---

[a] Taken to be 22.9% of the industrial finfish catches based on the nationwide average.

Table E34. Gulf of St. Lawrence and Grand Banks Fisheries 1974 Commercial Catches[17,19,(a)]

A. Gulf of St. Lawrence[17,(a),(b)]

Type	Total Catch (10^6 kg)	Distributed Catch (kg/km^2-yr)
Finfish, Industry	Negligible	---
Finfish, Food	230.0	5.75×10^2
Crustaceans	24.0	6.00×10^1
Mollusks	17.5	4.4×10^1

B. Grand Banks[17,(a),(c),(d)]

Type	Total Catch (10^6 kg)	Distributed Catch (kg/km^2-yr)
Finfish, Industry	Negligible	---
Finfish, Food	2020	2245
Crustaceans	Not available	---
Mollusks	Not available	---

[a] Almost the entire commercial catch is Canadian and is intended for human consumption.

[b] Area $\simeq 4 \times 10^5$ km^2.

[c] Shelf Area[20] $\simeq 10^5$ km^2 (Labrador) + 4.9×10^5 km (New Foundland) + 3.1×10^5 km^2 (Nova Scotia) = 9.0×10^5 km^2.

[d] This is essentially ICNAF Areas 2, 3 and 4.[17] These areas do not match the continental shelf exactly. However, most of the catch comes from the shelf.

E4.4.1 Lower Mississippi River

The total commercial wild catch for the Mississippi River, all tributaries, and associated areas for 1973 was 34. x 10^6 kg of finfish and 2.7 x 10^6 kg of crustaceans (mostly crawfish).[11] Of this total, only a small part came from the Lower Mississippi River (Table E35).[11] (The Lower Mississippi is taken to be the region between Cairo, Illinois, and the Gulf of Mexico.)[20] In this region, some pond-raised finfish was also harvested. It is estimated that 10 percent of the pond-raised fish could possibly be contacted by Mississippi River water.* The recreational catch of finfish for the lower Mississippi is estimated to be equal to the commercial catch for that region. The total commercial and recreational catches for the Lower Mississippi are listed in Table E35. The corresponding average distributed catches are presented in Table E12.

Table E35. Riverine Fisheries Catches for 1973[11] (10^6 kg)

River	Commercial Catch	Recreational Catch	Total Catch
Lower Mississippi[(a)]			
Finfish, Industry	0.3	0.0	0.3
Finfish, Food	2.3[(b)]	2.2	6.5[(c)]
Crustaceans	0.05	0.0	0.05
Mollusks	---	---	---
Lower Missouri[(d)]			
Finfish, Food	0.17	0.15	0.32
Tennessee[(d)]			
Finfish, Food	3.92	9.76[(e)]	13.68
Total for These Rivers			
Finfish, Industry	0.30	0.0	0.30
Finfish, Food	6.39	12.11	20.50
Crustaceans	0.05	0.0	0.05
Mollusks	0.0	0.0	0.0

[a] Includes Mississippi between Cairo, Illinois, and the Gulf of Mexico (1600 km).[20]

[b] Excludes the pond-raised contribution of the region which is approximately 20 x 10^6 kg.

[c] Includes 10% of the total pond-raised contribution of the region.

[d] Industrial finfish catches, total crustacean catches and total mollusk catches are small relative to the finfish catch for human consumption.

[e] Estimated assuming the national average catch of finfish from small lakes and reservoirs of 4740 kg/(km^2-yr) (see Subsection E4.4.4) and a total surface area of 2059 km^2.[16]

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*Almost all of the pond-raised fish from the vicinity (20 x 10^6 kg/yr) come from the Atchafalaya and Gulf of Mexico drainage areas. Some water from the Mississippi River is diverted to the Atchafalaya watershed as a flood control measure during periods of peak flow. It is assumed for this study that diversion would not be carried out in the event of an accident resulting in major radioactive pollution of the Mississippi River. In the event of an accident coinciding with floodstage flow, it would be necessary to compare the risk of increased downstream flooding to the risk of polluting the Atchafalaya watershed; such a comparison is beyond the scope of this study.

E4.4.2 Lower Missouri River

The Missouri River catch is relatively small and most of it comes from the upstream reservoirs. The entire Missouri River drainage area had a commercial catch (1973) of 2.3×10^6 kg of finfish.[11] However, only 0.17×10^6 kg of this amount came from the lower states of Kansas and Missouri. It is not known how much of this comes from the mainstream and how much from tributaries. If it is conservatively assumed that the entire amount comes from the mainstream, and that the recreational catch is 0.15×10^6 kg, then the total catches for the lower Missouri are as listed in Table E35. The corresponding average distributed catches are as given in Table E12.

E4.4.3 Tennessee River

The entire Tennessee River drainage area had a commercial catch (1973) of 3.92×10^6 kg.[11] The tributaries of the Tennessee are rather small and narrow and the free-flowing stretches of the Tennessee are rather short; consequently, the Tennessee River catch probably comes primarily from the Tennessee River reservoirs. The recreational catch from the Tennessee is not known; an estimate can be made from the national average for small lakes, reservoirs and man-made ponds (see Subsection E4.4.4). The resulting total catches are summarized in Table E35 and the total corresponding total annual distributed catches are given in Table E12.

E4.4.4 Other Rivers

The total finfish harvest from all rivers and streams is not given in the data considered. However, it can be approximated from the total freshwater finfish harvest.

The total recreational finfish harvest from all lakes, reservoirs and rivers in the U. S., excluding the Great Lakes, was estimated to be approximately 350×10^6 kg in 1970.[21] Because of a larger population, more leisure time, and a higher per capita income, the present freshwater harvest is estimated to be approximately 600×10^6 kg.*

The Sport Fishing Institute[14] estimated that the recreational harvest from lakes and reservoirs was about 1700 kg/km^2, and that the harvest from streams and rivers was about 500 kg/km^2, that is, about a factor of three lower. If it can be assumed that the ratio of the harvest from lakes and reservoirs to that from rivers and streams is correct, then the present harvest from all rivers is approximately 32×10^6 kg/yr and that from all lakes and rivers is 568×10^6 kg/yr.* The corresponding average distributed catches for rivers and streams, as well as those for small lakes and reservoirs, are given in Table E12.**,[16]

The freshwater commercial harvest outside the Great Lakes and the Mississippi River and its tributaries is negligible. Therefore, for most of the rivers in the United States, the total catch is essentially equal to the recreational catch.

E5 Irrigated Food

Almost all of the irrigation in the U. S. occurs in Louisiana and in the states west of the Mississippi (Table E36 and Figure E5).[23] In contrast, the majority of the nuclear power plants are located east of the Mississippi (Figure 4.2 in Chapter 4). Consequently, irrigation is not usually a significant pathway. However, the irrigation pathway can be important in certain situations.

The irrigation pathway to the human population includes the ingestion of irrigated crops, plus that of animal products and by-products derived from farm animals fed with irrigated crops and/or "irrigation" water. Just as for the aquatic-food pathway, little information is available on the actual individual assumption rates and affected population sizes for the irrigated-food pathway. Instead, the data which is most readily obtainable is that required for estimating the magnitude of the irrigated-food harvests themselves. The parameters required to calculate the irrigation pathway contribution to total population dose by considering the irrigated-food harvests include the following: the volumes of water used for irrigation; the types of irrigation utilized; the rates of water application employed; and the types of crops grown.

The irrigation volumes presented in this appendix are for all the water initially taken from various waterbodies for the purpose of irrigation. These volumes do not include the losses incurred between the source and the location of the irrigation. Nationally, conveyance losses alone average approximately 30 percent of the water withdrawn for irrigation. Such losses are ignored in this study. Summary irrigation data for the 18 individual Water Resource Regions (WRR's) of the contiguous U. S., shown in Figure E5, are presented in Table E36.[23]

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*There were actually twice as many angler-days in 1975 as in 1970,[21,22] however, it is expected that overfishing will slighly reduce the catch, so that the 1975 harvest is taken to be less than double the 1970 harvest.

*The surface area of all rivers is 2×10^4 km^2 and that of all lakes and reservoirs is 12×10^4 km^2.

**The total length of all rivers and streams is taken to be two times the sum of the lengths of all the major rivers given in Reference 15 (7.0×10^4 km).

Table E36. Irrigation Data for Water Resources Regions (1969)[23]

Region Number	Water Resources Region[(a)]	Volume of Irrigation Water[(b)] (m³/yr)	Percent from Surface Water[(b)(c)]	Sprinkling Fraction[(c)]
1	New England	3.1×10^7	90	79
2	Middle Atlantic	1.1×10^8	51	95
3	South Atlantic - Gulf	1.4×10^9	56	35
4	Great Lakes	9.8×10^7	48	97
5	Ohio	3.1×10^7	65	94
6	Tennessee	2.4×10^7	30	33
7	Upper Mississippi	1.1×10^8	30	89
8	Lower Mississippi	2.9×10^9	28	1
9	Souris-Red-Rainy	2.4×10^7	79	22
10	Missouri	1.4×10^{10}	62	8
11	Arkansas-White-Red	7.9×10^9	16	9
12	Texas - Gulf	7.9×10^9	18	15
13	Rio Grande	4.1×10^9	58	1
14	Upper Colorado	4.1×10^9	96	1
15	Lower Colorado	6.2×10^9	48	1
16	Great Basin	4.3×10^9	79	4
17	Columbia-North Pacific	1.8×10^{10}	81	26
18	California-South Pacific	2.6×10^{10}	61	11

[a] These regions are depicted in Figure E5.

[b] Table 13, 1969 Census of Agriculture.[23]

[c] 100 minus % from surface water = % from groundwater.

[d] Calculated using Table 8, 1969 Census of Agriculture.[23]

All irrigation methods can result in contamination of crops via root uptake of radionuclides present in the irrigation water. In addition, sprinkling can result in the direct deposition of radionuclides onto crops. The average fraction of irrigation water applied through sprinkling is given in Table E36 for each WRR.

The irrigation dose depends not only upon the irrigation volumes and the types of irrigation, but also upon the crops which are irrigated and upon their yields. Weighted averages of crop yields for the various irrigation regions considered in this study are given in Table E37,[23] along with national averages for all farmland.

The irrigation dose also depends upon how the irrigated crops are ultimately utilized. The destination of crops in each region—whether intended for human consumption or animal feeding or industrial use—is not known. However, national averages are available and are given in Table E38.[9]

The irrigation dose received via the portion of the pathway involving farm animals depends upon both the feeding characteristics of the animals and the production of food for humans by the animals. Tables E39[9] and E40[9] show a breakdown of feeds consumed by all farm animals; the proportions of each type of feed vary with the type of animal. Part A of Table E41[9] shows the annual production of meat while Part B of that table shows annual production of milk and eggs.

The irrigation pathway is not addressed for the generic ocean and estuary sites.

E5.1 Dry Sites

Power plant sites are listed as "dry" sites only if the average groundwater travel time to surface water is very long. Most of the sites listed as "dry" sites in Table E1 were so categorized only because they have a very low groundwater velocity. Only three dry sites have both reasonably long distances to travel to surface water, and hence potentially irrigable land between the site and the nearest surface water, and relatively large groundwater velocities. This sample is small enough for averaging, and so each site is considered in detail in the addendum of this appendix. In general, it can be said that the groundwater irrigation pathway at dry sites is not very significant.

Table E37. Average Crop Yields for Irrigated Land (in MT/km^2 for yields as harvested)[23]

Crop	US Average(a)	Great Lakes	Arizona	California	Tennessee Valley	Lower Mississippi
Forage(b)	(c)	1698	1365	1412	1570	1862
Corn for grain	575	626	452	596	383	525
Rice	481	(d)	(d)	632	(d)	536
Cottonseed	92	(e)	(e)	(e)	(e)	(e)
Grain(f)	(c)	336	473	415	(d)	437
Soybeans	187	160	(d)	(d)	(d)	175
Dry beans	136	162	137	194	(d)	(d)
Vegetables	1511	1163	1925	2811	690	863
Fruits	1758	(e)	(e)	(e)	(e)	(e)
Sugar cane	8410	(d)	(d)	(d)	(d)	4947
Sugar beets	4520	3897	4068	4682	(d)	(d)

[a] US averages for all land.

[b] Alfalfa, clover, timothy, wild hay, corn for silage.

[c] Average depends on mix of crops grown.

[d] Not grown, or too small to be meaningful.

[e] Not available.

[f] Wheat, barley, oats, rye, sorghym grain.

Table E38. Destination of All U.S. Crops (1973 data)[9]

Crop	Supply	Units	Quantity Consumed by Humans	Quantity Consumed by Animals	Human (%)	Animal (%)
Forage	154	10^6 tons	(a)	130	0	84
Corn for Grain	6353	10^6 bushels	1650(c)	4310	26	68
Rice	98701	10^3 cwt	54086(c)	442	55	~0
Cottonseed	5393	10^3 tons	122	2238(c)	2	41
Grain:	2151	10^6 bushel	1710	177	79	12
Wheat	60	"	15	19	25	32
Oats	1076	"	116(c)	709	11	66
Barley	602	"	74(c)	238	12	40
Sorghum	1010	"	4	1872	~0	86
All Grain(d)	4899	"	1919	2015	39	41
Soy beans	48785	10^3 tons	1805	27090(c)	4	56
Vegetables	22430(e)	10^3 tons	(b)	(a)	100	0
Fruits	24997(e)	"	(b)	(a)	100	0
Sugar cane	26584(e)	"	2719(f)	(g)	10	(g)
Sugar beets	24540(e)	"	3000(h)	1575	12	6

Note: Percentages will not necessarily add to 100; remainder of crop is wasted, lost in preparation, or used for industrial purposes.

[a] Assumed zero.

[b] Assumed all.

[c] Includes all exports.

[d] All grain other than corn.

[e] Production, not supply.

[f] Sugar, raw equivalent.

[g] Some bagasse is fed to cattle; the quantity could not be found.

[h] Refined sugar equivalent.

[i] Dry pulp. Some wet pulp is also fed, the amount is not known.

Table E39. Domestic Consumption of Feed by Cattle and Poultry for 1972[9]

Feed	Consumption (in metric tonnes)	Percent of Diet
Grain	152.4×10^6	27.9
Soybean Meal	10.9×10^6	2.0
Cottonseed Meal	2.0×10^6	0.3
Other Oil Meal	0.5×10^6	0.1
Fish Meal	0.5×10^6	0.1
Meat Meal	1.6×10^6	0.3
Dried Milk	0.3×10^6	0.1
Beet Pulp	1.4×10^6	0.2
Alfalfa Meal	1.6×10^6	0.3
Corn silage	98.6×10^6	18.0
Hay	118.0×10^6	21.6
Pasture	159.0×10^6	29.1
Total	546.8×10^6	100.0

Table E40. Feed Consumed by Farm Animals in 1972, per Unit of Production Quantity of Feed Expressed as Equivalent Feeding Value of Corn[9]

Type of Animal	Consumption (kg)	Units
Dairy Cattle	1.11	Per kg Milk
Beef Cattle	11.42	Per kg live weight
Sheep and Lambs	18.14	Per kg live weight
Hens	0.26	Per egg
Broilers	2.43	Per kg live weight
Turkeys	5.15	Per kg live weight
Hogs	6.33	Per kg live weight

Source: Agricultural Statistics, 1974.[9]

Table E41. Total Meat, Egg and Milk Production for 1973[9]

A. Meat Production[a]

Type	Meat Weight (10^6 kg)	Average Live Weight (kg)	Animals Slaughtered (thousand head)	Average Meat Weight (kg)
Beef	9670	480	34,027	284
Pork	5800	110	77,846	75
Lamb and Mutton	234	49	9,798	24
Veal	162	94	2,376	68
Lard	570	-	(all animals)	-
Broilers & Chickens	4053	1.74	3,237,523	1.25
Turkeys	890	8.41	132,153	6.73

B. Egg and Milk Production

Animals	Head	Yearly Production	Production/Head
Layers	2.928×10^8	66.6×10^9 eggs	227 eggs/head-yr
Milk Cows[b]	1.162×10^7	52.56×10^9 kg milk	3391 kg/head-yr
--[c]	3.874×10^6		

Source: Agricultural Statistics, 1974.[9]

[a] Includes meat consumed on farm where raised.

[b] Number that have already calved.

[c] Heifers 500 lb. and over, for replacement.

The irrigation rates, the sprinkling fractions and the irrigation depths for the three sites are summarized in Table E42;[23] average values appropriate for a "generic" dry site are also given in that table. Specific crop percentages assumed to be appropriate for land surrounding the three sites are presented in Table 43[23] and summarized in Table E44.

Table E42. Irrigation Rates, Sprinkling Fractions, and Irrigation Depths[23]

Sites	Irrigation Volume (m^3/yr)	Irrigation Rate (m^3/km^2)[a]	Irrigation Rate (m^3/km^2)[b]	Sprinkling Fraction	Average Irrigation Depth (m/yr)[c]
Dry Sites					
1	-	1.30×10^6	6.6×10^4	0.01	1.30
2	-	1.09×10^6	6.5×10^4	0.14	1.09
3	-	1.45×10^6	0	-	-
Average of 3	-	1.28×10^6	4.4×10^4	0.05	1.20
Average of 17[d]	-	1.40×10^5	7.6×10^3	0.02	0.14
Great Lakes	7.0×10^6	1.7×10^5	8.91×10^2	0.97	0.17
Rivers					
Tennessee-Ohio	3.68×10^6	2.6×10^5	2.2×10^4	0.33	0.26
Lower Missouri	-	Negligible	Negligible	-	-
Lower Mississippi	8.14×10^7	3.9×10^5	2.3×10^4	0.01	0.39

[a] Average over all irrigated land; total rate is given.

[b] Averaged over all land; groundwater rate is given for "dry" sites and surface water rate is given for Great Lakes and river sites.

[c] Applies only to irrigated land.

[d] Only two of the above sites were included in the RSS and are included in this average.

Table E43. Percentage Use of Irrigated Land[23]

	Dry Sites	Great Lakes Sites	Tennessee River Sites	Lower Mississippi Sites
Pasture	3.1	--	0.5	0.1
Harvested forage crops	15.1	2.9	7.9	0.2
Corn for grain	1.4	11.5	5.3	1.6
Rice, etc.	3.3	--	--	61.7
Other grains	19.9	3.3	3.5	0.8
Cotton	15.6	--	35.0	9.5
Soybeans	--	3.3	2.0	24.3
Dry beans	16.8	2.9	--	--
Vegetables	9.5	60.0	32.4	0.4
Fruits, nuts, berries	13.1	15.3	2.7	0.3
Sugar cane	--	--	--	1.1
Sugar beets	3.2	0.8	--	--
Tobacco and other	--	--	10.7	--

Table E44. General-Site Crop Disposition

Disposition (fraction)	Dry	Great Lakes	Tennessee-Ohio	Lower Mississippi
forage(a)	.182	.029	.084	.003
soybeans(a)	.000	.033	.020	.243
inedible(b) (cotton, tobacco, ...)	.156	.000	.457	.095
grain(c),(d)	.246	.148	.088	.641(d)
vegetables, fruits, nuts(e)	.227	.753	.351	.007
miscellaneous(e)	.194	.037	.020	.011

Approximate yields for each general class of crops are taken to be the same for all regions; the values used are 1.5 kg/m^2 for forage, 0.2 kg/m^2 for soybeans, 0.5 kg/m^2 for grain, 1.5 kg/m^2 for vegetables, fruits and nuts, and 0.5 kg/m^2 for miscellaneous.

[a] Assumed to be ingested entirely by farm animals.

[b] Assumed to be ingested by neither humans or farm animals.

[c] Assumed to be ingested 50% by humans and 50% by animals, except for the Lower Mississippi.

[d] The grain irrigated along the Lower Mississippi is almost entirely rice; therefore, essentially 100% of all grain grown there is consumed by humans.

[e] Assumed to be ingested entirely by humans.

E5.2 Great Lakes Sites

The total annual irrigation volume used in the Great Lakes Water Resources Region is 9.8 x 10^7 m^3/yr (Table E36). Forty-eight percent of that volume is taken from surface water. It is assumed that one-third of the irrigation water taken from surface water in the Great Lakes WRR comes from the Great Lakes themselves. Distributing one-third of the surface water irrigation volume over the lakes in proportion to the surrounding U. S. population (see Subsection E3.2 and Table E3) results in the estimated volumes listed in Table E45.

Canadian irrigation must also be considered. The Canadian shorelines of Lakes Superior and Huron have very small populations and little agriculture. However, the Canadian shores of Lakes Erie and Ontario are heavily settled, and additional irrigation water could be used there. For these two lakes, it is assumed that irrigation usage along the U. S. and the Canadian shores is proportional to the shoreline lengths. The resulting total irrigation volumes are given in Table E45. The irrigation volumes as functions of shoreline length are also given in that table.

Table E45. Estimated Irrigation Volumes for Great Lakes Sites

Lake	U.S. Volume (m^3/yr)	Total Volume (m^3/yr)		Distributed Usage(a) (m^3/km-yr)
Superior	2.0 x 10^5	2.5 x 10^5		137
Michigan	7.0 x 10^6	7.0 x 10^6		2624
Huron	5.0 x 10^5	5.0 x 10^5		551
Erie	6.0 x 10^6	1.2 x 10^7		8780
Ontario	1.3 x 10^6	3.1 x 10^6		2756
	Total	12.3 x 10^7	Average	1803

[a] Distributed along the shoreline.

The average irrigation rates both for land utilizing irrigation and for all farmland in the Great Lakes WRR are given in Table E42. A comparison of the two rates indicates that irrigation is not used for most farmland in the region. The sprinkling fraction for the region is also given in Table E42.

Most of the irrigation water in this region is used for crops consumed by humans (Table E43). The largest fraction of the acreage is used for vegetables, fruits, and corn (for grain). Only small fractions are used for other crops. The irrigated land usage is summarized in Table E44.

E5.3 River System Sites

The three river systems considered in this study are the Tennessee-Ohio, the Lower Missouri, and the Lower Mississippi. Each of the areas has such different land use and irrigation practices that they are considered separately.

Only the irrigation water removed from the mainstream of each river requires consideration. The water in tributaries would not be polluted by an accident unless the accident site were on a tributary; in such a case, just the single tributary (and all its lower reaches) would be affected. However, the information about the water removed from each tributary and the mainstream is not normally available. It has been assumed in obtaining estimates of irrigation volumes that the amount of water taken from any section of a river system is just proportional to the length of the section.

E5.3.1 Tennessee-Ohio River System

The Tennessee and its tributaries are included in WRR 06 (Figure E7). Thirty percent of the water used for irrigation in this region comes from surface sources. The total amount of irrigation water used in WRR 06 is 2.45×10^7 m^3 so that approximately 7.36×10^6 m^3, comes from the surface water sources. The length of the mainstream is roughly equal to the lengths of the major tributaries so that 3.68×10^6 m^3 can be assumed to come from the mainstream. The length of the Tennessee River from Watt's Bar Dam to the mouth is 852 km, so the average usage distributed over the entire length of the mainstream and its reservoirs is 4320 m^3/km.

The average irrigation rates both for land using irrigation and for all farmland in the Tennessee WRR are given in Table E42. A comparison of the two rates indicated that some irrigation is used on most of the farmland in the region. The sprinkling fraction for the region is given in Table E42.

The percentages of the irrigated area devoted to various crops are shown in Table E43. Almost half of the irrigated land in this region is used to grow essentially non-edible crops. The irrigated land usage is summarized in Table E44.

E5.3.2 Lower Missouri River

The Missouri and its tributaries are included in WRR 10 (Figure E7). Irrigation water usage from the Missouri and its tributaries is extensive. However, essentially 100 percent of all water used for irrigation is taken upstream from any built or planned reactor. In the absence of specific information to the contrary, it is assumed that nearly all irrigation from surface water sources uses water from upstream reaches.

E5.3.3 Lower Mississippi River

The Lower Mississippi River is located in WRR 08 (Figures E5 and E7). This region also includes short lengths of some major tributaries to the Mississippi, plus the Atchafalaya River and the Gulf of Mexico drainage areas. A total of 2.93×10^9 m^3 per year is used for irrigation in WRR 08. The total irrigation water usage from surface sources in the WRR is 5.9×10^8 m^3/yr. The portion of that irrigation water usage which is taken from the Lower Mississippi is estimated to be 8.1×10^7 m^3/yr (see Table E46). The length of the Lower Mississippi River is approximately 1600 km, so that the average irrigation water usage is 5.1×10^4 m^3/km.

The average irrigation rates both for just irrigated land and for all farmland in the vicinity of the lower Mississippi are given in Table E42. A comparison of the two rates indicates that most farmland in the region is not irrigated. The sprinkling fraction for the region is also given in Table E42.

The nature of cropping changes from North to South. However, for simplicity, percentage usage for the entire Lower Mississippi Water Resources Region has been used. These percentages are given in Table E43. Almost two-thirds of the irrigated land in this region is used to grow rice. The irrigated land usage is summarized in Table E44.

E6 Shoreline Usage

The shoreline pathway contribution to total dose depends on the estimated population exposure. Shoreline exposure is assumed to occur primarily as a result of water-related recreational activities, and is estimated from available boating, fishing, swimming, and waterfowl hunting data.[21,22,24,25] All exposure is assumed to be uniformly distributed throughout the year. The shoreline pathway is not considered for the generic dry site.

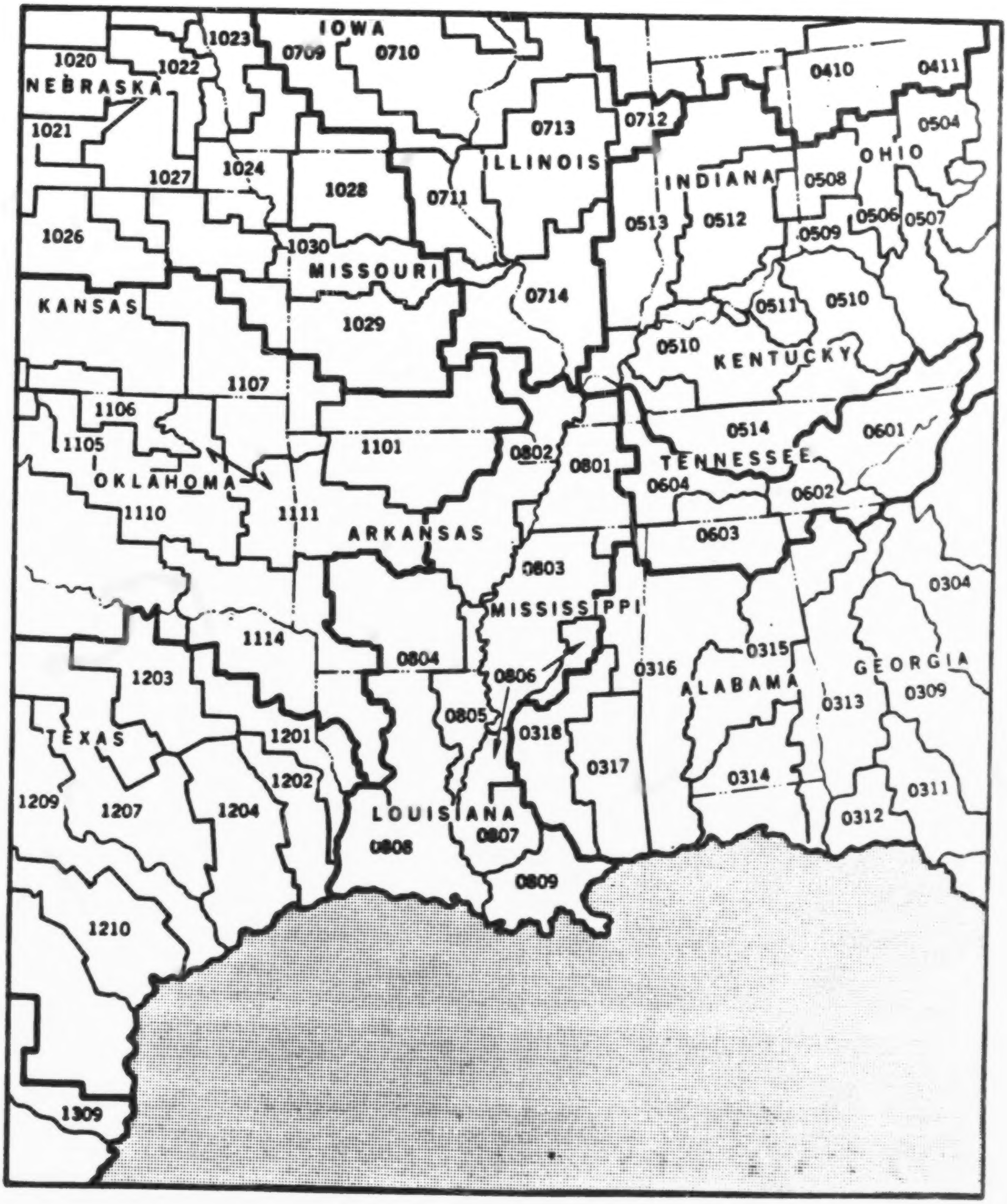
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Figure E7. Water Resource Regions and Subregions Around the Lower Mississippi

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Table E46. Fractions and Total Amounts of Irrigation Water from the Lower Mississippi River Mainstream

	WRS	fSA(a)	fMS(b)	fMiss(c)	Total Volume (m³/yr)(d)
Louisiana	0807	.15	.5	.12	3.90×10^7
	0805 (part)	.15	.3		
Mississippi	0806	.1	.3	.12	5.99×10^6
	0803	.3	.3		
Arkansas	0805 (part)	.05	.9	.14	3.43×10^7
	0802 (part)	.3	.3		
Tennessee(e)				0.50	2.16×10^6
				Total	8.14×10^7

[a] f_{SA} is the estimated ratio of the state's area in each WRS that includes the Mississippi drainage system to the total area of the state within WRR08.

[b] f_{MS} is the ratio of the Mississippi mainstream length to the total major stream length in the subareas in the state.

[c] $$f^{state}_{Miss} = \sum_{subareas} f_{SA} \cdot f_{MS} .$$

[d] $$\text{total volume} = \sum_{states} f^{state}_{Miss} \cdot V_{state} ,$$

where

V_{state} is the surface water irrigation volume for the state.

[e] WRS 0801 contains a relatively small fraction of the total area of the state. However, because it encompasses a very valuable agricultural area, the fraction for the Mississippi River has been made conservatively high.

A number of very rough, although probably conservative, approximations have been made in this section and in the following section on immersion; such approximations are justified for cases in which the contribution from the considered pathway is small compared to the contributions from other pathways. For calculating the exposure for specific sites, these approximations should not be used.

Recreational data have been taken extensively from Reference 21. References 24 and 25 have also been utilized. The data available for estimating the total shoreline population usages differ by as much as a factor of two (Tables E47 and E48). The arbitrary distribution of the total shoreline usages among various types of waterbodies can result in an even larger difference between estimates.

Recreational usage is measured in terms of person-days of activity. These values can be readily converted to the number of hours of actual exposure if the average numbers of hours per day per person of activity are known. The following assumptions are made about these averages in this section: [25] a day of boating involves 1 hour of shoreline activity; a day of swimming, 2.5 hours; a day of fishing, 3 hours; a day of waterfowl hunting, 6 hours; and a day of walking, 2 hours.

From this information and the population usage data presented in Tables E47 and E48, the total U. S. recreational shoreline usage is estimated to be 7.56×10^9 person-hrs/yr. The details are given in Table E49.

The shoreline usage resulting from all commercial activities is arbitarily taken to be 25 percent of the total recreational shoreline usage, that is, 1.89×10^9 person-hrs/yr. As a result, the total shoreline usage for all activities is approximately 9.45×10^9 person-hrs/yr.

The distribution of the total shoreline usage with respect to the type of nearest waterbody is considered in the next four subsections. The results are summarized in Table E50.

Table E47. 1970 U.S. Water Contact Recreation (Other than Fishing) by Persons of Age Nine or Over

	Participants[21] (10^6)	Person-days[21] (10^6)	Average Days Per Person	Range of Person-days Data (10^6)[(a)]
Boating, Sailing, Canoeing	41	420	10.2	212[22] - 684[16,(b)]
Swimming	77	1,700	22.1	744[22,(c)] - 2323[16,(b)]
Freshwater Angling[(d)]	29	590	17.9	<278[22,(e)] - 1130[24]
Saltwater Angling	9.5	110	11.6	<110[21] - 242[24]
Waterfowl Hunting	3	25	3.6	25[21] - 56[24]

[a] Projected recreational participation data tend to increase much more rapidly than the population itself; therefore, a part of this variation is due to using estimates for different years.

[b] This is an estimated value for 1975.

[c] This is only for the summer months.

[d] The number of anglers includes only those who spent $7.50 or more, or who reported three or more angler-days during the 1970 Census survey.

[e] This includes both saltwater and freshwater fishing, but is only for the summer months.

Table E48. Water Contact Recreation in the U.S. (1972)

	Summer Person-days[25,(a)]	Hours/ Person-day[25,(a)]	Estimated Person-days/yr[(b)]
Boating, Sailing, Canoeing, Waterskiing	231.0	2.9	462.0
Swimming[(c)]	744.1	2.7	1488.0
Fishing	278.2	4.4	556.4
Walking	496.3	1.9	98.2[(d)]

[a] This is taken from the National Recreation Survey for the Summer of 1972;[25] it includes all participants over 12 years of age.

[b] Year-round activity is assumed to be twice the summer activity.

[c] This includes swimming in pools and in natural waterbodies.

[d] 10 percent of all walking for pleasure is assumed to occur along shorelines and flood plains.

Table E49. Recreational Shoreline Explosure[a]

Recreational Activity	Person-hours/ Person-day	Person-days/yr	Person-hrs/yr
Boating	1	4.2×10^8	0.42×10^9
Swimming	2.5	1.7×10^9	4.25×10^9
Fishing	3	6.3×10^8	1.89×10^9
Waterfowl hunting	6	2.5×10^7	0.15×10^9
Walking	2	4.0×10^8	0.80×10^9
Total			7.56×10^9

[a]The total shoreline exposure is taken to be 1.25 times the recreational shoreline exposure.

Table E50. Estimated Population Usages for the Shoreline and Immersion Pathways

	Shoreline		Immersion	
Waterbody	Total Person-hr/yr	Distributed person-hr/(km-yr)	Total person-hr/yr	Distributed person-hr/(km-yr)
Great Lake	9.1×10^8	-	4.5×10^8	-
Superior	1.9×10^7	3.9×10^3	9.3×10^6	1.9×10^3
Michigan	3.5×10^8	1.3×10^5	1.8×10^8	6.4×10^4
Huron	4.4×10^7	1.6×10^4	2.1×10^7	8.1×10^3
Erie	3.5×10^8	2.5×10^5	1.8×10^8	1.2×10^5
Ontario	1.6×10^8	1.4×10^5	7.8×10^7	7.0×10^4
Estuary	2.3×10^4	2.8×10^4[a]	9.8×10^8	1.2×10^4[a]
Ocean	7.5×10^8	1.0×10^5	3.3×10^8	4.4×10^4
River System	3.8×10^9	2.6×10^4	1.8×10^9	1.3×10^4
Total	9.5×10^9			

[a]Based on tidal shoreline length.

E6.1 Great Lakes Sites

Only a few percent of the total U. S. recreational freshwater fish harvest is from the Great Lakes. However, the Great Lakes region has approximately 10 to 15 percent of the total U. S. population. In agreement with these, it is assumed that 10 percent of the total U. S. boating, swimming, freshwater fishing and waterfowl hunting activity occurs on the Great Lakes. (The Canadian shoreline usage is assumed to be included in this estimate).

This approximation for the Great Lakes shoreline usage gives a total population usage of 9.1×10^8 person-hrs/yr. Distributing this value over the individual lakes in proportion to the surrounding drinking-water population (Table E3) results in the following: Superior, 1.88×10^7 person-hrs/yr; Michigan, 3.50×10^8 person-hrs/yr; Huron, 4.4×10^7 person-hrs/yr; Erie, 3.5×10^8 person-hrs/yr; and Ontario, 1.58×10^8 person-hrs/yr. Dividing the value for the total usage for each lake by the corresponding shoreline length results in an average usage of 3.9×10^3 person-hrs/yr-km for Lake Superior; 1.3×10^5 person-hrs/yr-km for Lake Michigan; 1.6×10^4 person-hrs/yr-km for Lake Huron; 2.5×10^5 person-hrs/yr-km for Lake Erie; and, 1.4×10^5 person-hrs/yr-km for Lake Ontario.

E6.2 Estuarine Sites

It is assumed that 30 percent of the total U. S. boating, swimming, saltwater fishing, and waterfowl hunting occurs in estuarial regions. Therefore, the total shoreline usage for tidal regions is approximately 2.3×10^9 person-hrs/yr. Dividing this by the total U. S. tidal shoreline length (8.2×10^4 km) results in an average usage of 2.8×10^4 person-hrs/yr-(km of tidal shoreline).

E6.3 Ocean Sites

It is assumed that 10 percent of the total U.S. boating, swimming, saltwater fishing and waterfowl hunting occurs along coastal shorelines of oceans. The total shoreline usage along oceans is therefore 7.5 x 10^8 person-hrs/yr. The resulting average population shoreline usage is 1.0 x 10^5 person-hrs/(km of coastal shoreline)-yr.

E6.4 River System Sites

Assuming that 40 percent of the total U.S. boating, swimming, freshwater fishing and waterfowl hunting occurs along river system shorelines results in a total shoreline usage of 3.8 x 10^9 person-hrs/yr. Dividing the total usage by the total surface area of all major rivers and streams in the 48 contiguous states (2.3 x 10^4 km^2)[16] results in an average usage of 1.6 x 10^5 person-hrs/km^2-yr; dividing the total usage by the total length of all major rivers and streams (1.4 x 10^5 km)[16,*] results in 2.6 x 10^4 person-hrs/(km-yr).

E7 Immersion

The swimming pathway contribution to total dose depends on the estimated population exposure. The pathway includes immersion in the water (swimming) and boating or other recreational activities on the surface of the water. Those activities on the surface are assumed to result in 1/2 the exposure of swimming (2π rather than 4π irradiation). All usage values are assumed to be uniformly distributed throughout the year. The swimming pathway is not considered for the generic dry site.

Recreational usage measured in terms of person-days of activity can be converted to the number of hours of actual exposure by using the average number of hours per day per person of activity. The following assumptions are made in this section:[25] a day of boating involves 3 hours of immersion-related activity; a day of swimming, 1 hour; a day of fishing, 4 hours; and a day of waterfowl hunting, 6 hours. From this information and the population usage data presented in Tables E47 and E48, the total U.S. recreational immersion usage is estimated to be 3.81 x 10^9 person-hrs/yr. The details are given in Table E51.

The immersion usage resulting from all commercial activities is arbitrarily taken to be 25 percent of the total recreational immersion usage, that is, 0.95 x 10^6 person-hrs/yr. As a result, the total immersion usage from all activities is approximately 4.76 x 10^9 person-hrs/yr.

The distribution of the total immersion usage with respect to the type of nearest waterbody is considered in the next four subsections. The results are summarized in Table E50.

E7.1 Great Lakes Sites

It is again assumed that 10 percent of all the U.S. shoreline and immersion-related activity (excluding saltwater fishing) occurs on the Great Lakes. Therefore, the combined effective immersion usage for the Great Lakes is 4.5 x 10^8 person-hours/yr. Distributing this combined usage to the individual lakes in proportion to the surrounding drinking water populations provides the following population usages: Lake Superior, 9.3 x 10^6 person-hrs/yr; Lake Michigan, 1.8 x 10^6 person-hrs/yr; Huron, 2.1 x 10^7 person-hrs/yr; Erie, 1.8 x 10^8 person-hrs/yr; and Ontario, 7.8 x 10^7 person-hrs/yr. Dividing the total usage of each lake by the corresponding shoreline length results in average usage of 1.9 x 10^3 person-hrs/km-yr for Superior; 6.4 x 10^4 person-hrs/km-yr for Michigan; 8.1 x 10^3 person-hrs/km-yr for Huron; 1.2 x 10^5 person-hrs/km-yr for Erie; and 7.0 x 10^4 person-hrs-yr for Ontario.

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[*] The total length of all rivers and streams is taken to be two times the sum of the lengths of all the major rivers given in Reference 16 (7.0 x 10^4 km).

Table E51. Recreational Immersion Exposure[a]

Recreational Activity	Person-hours/ Person-day	Person-days/yr	Person-hrs/yr
Boating	3	4.2 x 10^8	0.63 x 10^9
Swimming	1	1.7 x 10^9	1.70 x 10^9
Freshwater fishing	4	5.9 x 10^8	1.18 x 10^9
Saltwater fishing	4	1.1 x 10^8	0.22 x 10^9
Waterfowl hunting	6	2.5 x 10^7	0.08 x 10^9
Total			3.81 x 10^9

[a] The total immersion exposure is taken to be 1.25 times the recreational immersion exposure.

E7.2 Estuary Sites

As in the shoreline exposure discussion, it is assumed that 30 percent of the total U.S. shoreline and immersion-related usage (excluding freshwater fishing) takes place in estuarial regions. Consequently, the total effective immersion usage at estuaries is 9.8×10^8 person-hrs/yr. Dividing this by the total U.S. tidal shoreline gives an average tidal immersion usage of 1.2×10^4 person-hr/(km of tidal shoreline)-yr.

E7.3 Ocean Sites

It is assumed again that 10 percent of the total U.S. immersion-related usage (excluding freshwater fishing) occurs on ocean coasts. As a result, the total effective immersion usage along U.S. ocean coasts is 3.3×10^8 person-hrs/yr. Dividing this by the total coastal shoreline length results in an average coastal immersion usage of 4.4×10^4 person-hrs/(km of coastal shoreline)-hr.

E7.4 River System Sites

Assuming that 40 percent of the total U.S. immersion-related exposure (except for saltwater fishing) occurs along river systems results in a total immersion usage of 1.8×10^9 person-hrs/yr. Dividing this by the total surface area of all the major rivers and streams in the contiguous 48 states results in an average usage along river systems of 7.6×10^4 person-hrs/km^2-yr; dividing this by the net length of all major rivers and streams in the contiguous 48 states results in 1.3×10^4 person-hrs/km-yr.

E8 Generic Site Populations

The populations employed for each of the generic sites are given in Table E52. The data presented in the previous subsections can be used to estimate the appropriateness of the values chosen.

Table E52. Generic Site "Populations"

	"Dry"	Great Lake[a,e]	Estuary[b,e]	Ocean[c,e]	River System[d,e]
Drinking Water	31 persons/km² x 548 ℓ/(person-yr)	11 x 10^6 persons x 548 ℓ/(person-yr) or 1900 person/km x 548 ℓ/(person-yr)	— —	—	1351 persons/km x 548 ℓ/(person-yr)
Aquatic Foods					
Finfish[e]	—	5.1(16.5) x 10^6 kg/yr or 1740(5600) kg/km-yr	870(2800) kg/km²-yr	16,700(53,800) kg/(km of coastal shoreline-yr) from 0-5 km offshore 42,400(137,000) kg/km of coastal shoreline-yr) beyond 5 km	1810(5820) kg/km-yr (in general, 490(1580) kg/km²-yr for rivers and 1470(4740) kg/km²-yr for reservoirs)
Crustaceans[e]	—	0	2260(5010) kg/km²-yr	11,900(26,500) kg/(km of coastal shoreline-yr) from 0-5 km offshore 5,930(13,200) kg/(km of coastal shoreline-yr) beyond 5 km	6(14) kg/km-yr
Mollusks[e]	—	0	1445 kg/km²-yr	7,050(7,050) kg/(km of coastal shoreline-yr) from 0-5 km offshore 22,200(22,200) kg/(km of coastal shoreline-yr) beyond 5 km	negligible
Irrigated Foods	7.6 x 10^3 m³/km²-yr (to a depth of 0.14 m)	7.0 x 10^6 m³/yr (to a depth of .17 m) or 3.1 x 10^3 m³/km-yr	—	—	3.47 x 10^4 m³/km[f] (to a depth of .39 m/yr)
Shoreline	—	3.50 x 10^8 person-hr/yr or 1.3 x 10^5 person-hrs/yr-km	2.8 x 10^4 person hrs/(yr-km of tidal shoreline) x 2 x 5[g,h,i]	1.0 x 10^5 person-hrs/(yr-km of coastal shoreline)	1.6 x 10^5 person-hrs/km²-yr; 2.6 x 10^4 person-hrs/km-yr
Immersion	—	1.80 x 10^8 person-hrs/yr or 6.4 x 10^4 person-hrs/yr-km	1.2 x 10^4 person-hrs/(yr-km of tidal shoreline) x 2 x 5[g,h,i]	4.4 x 10^4 person-hrs/(yr-km of coastal shoreline)	7.6 x 10^3 person-hrs/km²-yrs; 1.3 x 10^4 person-hrs/km-yr

[a] These populations are based on those of Lake Michigan.

[b] These populations are based on those for Chesapeake Bay.

[c] These populations are averages for the entire U.S. Atlantic, Gulf of Mexico, and Pacific Coasts.

[d] These populations are based on an average of the Tennessee, the Lower Missouri, and the Lower Mississippi Rivers.

[e] Amounts directly ingested by humans; amounts harvested for consumption are given in parentheses.

[f] This is averaged only over the Tennessee and the Lower Mississippi.

[g] Factor of two takes into account that an estuary has 2 shorelines.

[h] Arbitrary factor of five is to account for the fact that estuary shorelines tend to very irregular.

[i] The tidal shoreline includes bays, sounds, rivers and creeks to a point where the tidal water narrows to 100 feet. The coastal shoreline follows the apparent outline of the coast; the coastlines of sounds and bays are included only to a point where they narrow to 30 nautical miles.

Addendum E1

Irrigation Data for Dry Sites

E1.1 Site 1

This site is located in one of the most heavily cultivated parts of Water Resource Subarea (WRS) 1505. Essentially 100 percent of all cropland in the area is irrigated. However, only a fraction of the total land area is cropland. A local cropland fraction of 7.5 percent might be reasonable.

For the land actually irrigated in WRR 15, the average water application rate is 1.30 m^3/m^2 (1.3 x 10^6 m^3/km^2). If we assume 7.5 percent of land is cultivated, the application rate is 9.6 x 10^4 m^3/km^2 averaged over all land. The fraction of the irrigation water in this region which comes from the groundwater is approximately 0.69. Therefore, the application rate should be multiplied by 0.69 for the region before the groundwater reaches surface water. These irrigation rates are summarized in Table E42.[23]

Detailed specific crop percentages are known only for the entire state. These are presented in Table 43.[23] At least one-third of all the irrigated land in this state is used to grow crops not intended for human consumption. The irrigated land usage is summarized in Table E44.

E1.2 Site 2

This site lies in Water Resources Subarea 1805. For the entire state, 39 percent of all farmland and 83 percent of all harvested cropland is irrigated. This includes certain intensively cultivated regions in which all farmed land is irrigated. Of the total area of the state, 7.0 percent is irrigated. This is much too low for this site; it is assumed that essentially all nonurban land is cultivated, so that 39 percent of all land is assumed to be irrigated.

The average irrigation rate for WRS 1805 is 1.09 m^3/m^2 as compared with 0.9 m^3/m^2 for the entire state. The groundwater irrigation water usage is thus 0.425 m^3/m^2 (4.25 x 10^5 m^3/km^2) for all land near the site (1.09 m^3/m^2 for irrigated land, times 39 percent of farmland which is irrigated). The fraction of the irrigation water in this region which comes from groundwater is estimated to be 0.39. Therefore, the application rate should be multiplied by 0.39 before the groundwater reaches surface water. The irrigation rates are summarized in Table E42.

Specific cropland fractions are known only for the entire state, and may be in error as applied to this site. The state percentages for land which is completely irrigated are given in Table E43.[23] At least one-third of the irrigated land in this state is used to grow crops not intended for direct human consumption. The irrigated land usage is summarized in Table E44.

E1.3 Site 3

This site is located in WRS 1810. All farms in the vicinity are irrigated. However, essentially all the irrigation water comes from surface water. Consequently, groundwater usage for irrigation is probably not important in this region. Inasmuch as the groundwater travel time to the surface water is estimated to be relatively long (>10^6 days), most of the radioactivity will probably never reach any surface water at this site.

References

[1] U.S. Nuclear Regulatory Commission, *Reactor Safety Study*, "An Assessment of Accident Risks in U.S. Commercial Nuclear Power Plants," WASH-1400, NUREG-75/014, USNRC, Appendix VI, 1975.

[2] J. F. Fletcher, et al., *HERMES: A Digial Computer Code for Estimating Regional Radiological Effects from the Nuclear Power Industry*, HEDL-TME-71-168, Hanford Engineering, Hanford, Washington, 1971.

[3] U.S. Nuclear Regulatory Commission, "Calculations of Annual Doses to Man from Routine Releases of Reactor Effluents for the Purpose of Evaluating Compliance with 10 CFR Part 50, Appendix I," Regulatory Guide 1.109, USNRC, March 1976.

[4] International Commission on Radiological Protection, *Report of ICRP Committee II on Permissible Dose for Internal Radiation*, ICRP Publication 2, Pergamon Press, New York, 1959.

[5] Bureau of the Census, "1970 Census of Population, U.S. Summary: Number of Inhabitants," PC(1)A1, U.S. Department of Commerce, 1971.

[6] U.S. Nuclear Regulatory Commission, *Liquid Pathways Generic Study*, "Impacts of Accidental Radioactive Releases to the Hydrosphere from Floating and Land-Based Nuclear Power Plants," NUREG-0440, USNRC, Chapter 4, 1978.

[7] H. W. Gehm and J. I. Bergman, Editor, *Handbook of Water Resources and Pollution Control*, Van Nostrand Reinhold Co., 1976.

[8] National Marine Fisheries Service, "Fisheries of the United States," Current Fishery Statistics No. 6400, NOAA, 1973.

[9] United States Department of Agriculture, *Agricultural Statistics 1974*, USDA, 1974.

[10] National Marine Fisheries Service, *Fishery Statistics of the United States, 1974*, Statistical Digest No. 68, NOAA, 1975.

[11] National Marine Fisheries Service, *Fishery Statistics of the United States, 1973*, Statistical Digest No. 67, NOAA, 1974.

[12] National Marine Fisheries Service, *Fishery Statistics of the United States, 1971*, Statistical Digest No. 65, NOAA, 1972.

[13] L. H. Long, Editor, *The World Almanac and Book of Facts*, Newspaper Enterprise Association, New York, 1970.

[14] U.S. Nuclear Regulatory Commission, *Liquid Pathways Generic Study*, NUREG-0440, USNRC, Appendix D, 1978.

[15] S. A. Cain, "Estuarine Areas," Serial No. 90-3, U.S. House of Representatives, 28-76, 1967.

[16] D. K. Todd, Editor, *Water Encyclopedia*, Water Information Center, Inc., Huntington, NY, 1970.

[17] International Commission for the North Atlantic Fisheries, "Statistical Bulletin," Vol. 24, Dartmouth, Canada, 1976.

[18] B. E. Vaughn and J. A. Strand, "Biological Implications of a Marine Release of ^{90}Sr," Health Physics, *18*, 256, 1970.

[19] *Atlas of the Oceans*, Rand McNally Co., Chicago, IL, 1977.

[20] E. G. Struxness, et al., "Comprehensive Report of the Clinch River Study," ORNL 4035, ORNL, 1967.

[21] U.S. Fish and Wildlife Service, "1970 National Survey of Hunting and Fishing," U.S. Department of the Interior, 1974.

[22] U.S. Fish and Wildlife Service, "1975 National Survey of Hunting, Fishing, and Wildlife-Associated Activities," U.S. Department of the Interior, 1977.

[23] Bureau of the Census, *1969 Census of Agriculture, Irrigation*, U.S. Department of Commerce, Vol. IV, 1973.

[24] Bureau of the Census, *Statistical Abstract, 1977*, U.S. Department of Commerce, 1978.

[25] Bureau of Outdoor Recreation, *Outdoor Recreation, A Legacy for America*, U.S. Department of the Interior, Appendix A, 1973.

APPENDIX F

Interdiction

APPENDIX F

Interdiction

Authors: S. J. Niemczyk
W. B. Murfin
E. W. Appel

F1 Introduction

During a core-melt accident, radioactive material can be released directly to the atmosphere and/or to the subsoil. The human population can potentially be exposed to radiation from either of these types of releases. Certain mitigating actions can often be taken to lessen the exposure received by humans and other inhabitants of the environment. In this study, the term interdiction is taken to encompass all such actions which might be taken,[1,2] except for those emergency measures which might be taken within the first few days after the accident. (This definition of interdiction departs from that used in the Reactor Safety Study (RSS).[3] In that study, interdiction was taken to mean denial of the use of land and its improvements for a period of time.)

In this appendix, the mitigating actions which might be taken to lessen the radiation exposure received after a release of radioactivity to the hydrosphere are considered. In the rest of this section, the types of releases to the hydrosphere, the potential hydrospheric exposure pathways and the differences between the atmospheric and the hydrospheric releases are reviewed. In Section F2, interdictive procedures which can be taken close to the radioactive source (the site of the accident) are discussed. Various possible methods are reviewed, along with representative costs for implementing them. In Section F3, interdictive procedures which can be taken at some distance away from the accident site, along the exposure pathways to the human population, are dealt with. Both interruptive and restorative mitigating procedures are considered, along with costs which might be reasonable for implementing those procedures. In Sections F2 and F3, both source and pathway interdictive procedures are considered as functions of site characteristics. In the last section, F4, the available procedures and costs are summarized.

F1.1 Differences Between Atmospheric and Hydrospheric Pathways

The atmospheric pathway is the route taken by radioactive material upon release to the atmosphere following a reactor accident. In the case of a core-melt accident, radionuclides can be released into the atmosphere by either leakage from the containment through slight breaching of its integrity or else expulsion from the containment through massive rupture of the walls. Exposure can be either external, as from the radioactive cloud or from ground-deposited radionuclides, or internal, as from inhaled radioactivity or from contaminated food.

Probably the greatest threat posed by an atmospheric release is that its initial dispersion in the environment is relatively rapid. There is no known way to contain this type of release at its source, so that early exposure can be minimized only by very swiftly evacuating or sheltering the endangered people, and perhaps by issuing potassium iodide pills (to block the adsorption of inhaled radioiodines by the thyroid). However, there are several ways to minimize the adverse effects of an airborne release once it has been deposited out of the atmosphere. Initially, monitoring can be put into effect to determine concentrations of airborne material. This can be followed by procedures such as relocation of people, destruction of crops, restrictions and/or denial of land use, and decontamination of land and buildings.

In contrast, the hydrospheric pathway is that route taken by radioactive material released either into groundwater or surface water. In the case of a core-melt accident, radionuclides can be released to the hydrosphere in the following ways:

1. By release of highly contaminated sumpwater either through the containment base or

through some other path from the containment;

2. By gradual leaching into the groundwater from the cooled and solidified core debris; and,
3. By depressurization of the containment atmosphere into the subsoil.

Exposure can be either external, as from immersion or from shoreline-deposited radionuclides, or internal, as from ingestion of contaminated water and food.

Exposure via the hydrospheric pathways cannot occur before the radioactivity reaches a point of potential human contact, such as a drinking-water well or a surface waterbody. The travel of the radioactivity to such a point is generally through an aquifer or other such subterranean medium and consequently is relatively slow. As a result, immediate action following a hydrospheric release is not usually as critical as it is for an airborne release. However, in many cases interdiction might have to continue over a much longer period of time due to the relatively long residence times of at least some radionuclides in rather inaccessible portions of the hydrosphere; that is, after a release to the hydrosphere, transport of radionuclides into accessible waterbodies and subsequent deposition onto shorelines could continue over a period of many years. Consequently, certain interdictive procedures might need to be repeated periodically for releases into the hydrosphere. With this type of release, measures can usually be taken to prevent exposures without having to relocate the population.

Since time is often not as critical in preventing early exposure from a subsurface release, there are methods that can sometimes be employed to isolate the radioactivity at the source, such as injecting grout to form a barrier, pumping out contaminated water to permit treatment, and lowering the watertable to prevent contact of the water with the melt debris. Mitigation of the radiation hazard after the radioactivity has reached the groundwater or surface water might be possible also, for example, by restricting use of wells, confiscating contaminated fish, and dredging.

Releases to the hydrosphere are not necessarily initially to the ground. For example, contaminated sumpwater might be released essentially directly to surface water. Just as in the case of an atmospheric release, isolation of the radioactivity at the source would not be possible in such situations. In these cases, immediate action might be necessary to reduce the radiation hazard. Mitigating actions could include substituting drinking water and confiscating contaminated fish.

Whenever possible, interdiction in the vicinity of the radioactive source would usually be more effective than interdiction at points along the various exposure pathways. Not only would such source interdiction potentially be more effective, but it would also generally cost substantially less than pathway interdiction. In general, the costs for interdiction at the source for the hydrospheric pathways are expected to be less than the costs for interdiction along the exposure pathways for either the hydrospheric or the atmospheric releases.

The risks associated with the liquid pathway are thought to be less than those for the air pathway, since the feasibility of interdiction close to the source is generally much greater for the liquid pathway. Furthermore, the initial travel of the radioactivity through the ground to a location of potential exposure can take years. Consequently, even if no interdictive measures are employed, the amount of activity reaching the human environment by liquid pathways can be reduced significantly by radioactive decay.

In addition, most of the exposure via hydrospheric pathways can be regarded as being voluntary, whereas at least the early exposure via atmospheric pathways is comparatively involuntary. For example, whereas drinking water can be obtained from alternative sources relatively rapidly, substitute sources for air are not readily available. As a result, the direct consequences of a hydrospheric release can be reduced by avoidance and substitution, but the consequences resulting from at least early exposure to an atmospheric release often cannot be.

F1.2 Types of Releases

The relevant time scales for both atmospheric and hydrospheric releases are outlined in Table F1.

F2 Source Interdiction

F2.1 Methods of Source Interdiction

Possible methods to contain radioactive material within the area surrounding a reactor following a core-melt accident include the following:

1. injection or withdrawal of water;
2. lowering of the watertable;
3. installation of a grout curtain.[1]

Table F1. Release and Transport Characteristics

Type of Release	Duration of Release	Transport in Environment
Atmospheric	Hours to days.	Hours to days.
Hydrospheric:		
Melt debris	Weeks to years for groundwater contact; additional weeks to years for leaching.	Days to infinity, generally at least years.
Sumpwater	Hours to years.	Days to infinity, generally at least years, although rapid direct releases to accessible portions of the hydrosphere are possible.
Depressurization	Hours to months.	Days to infinity.

F2.1.1 Withdrawal and Injection of Water

Wells can be drilled in the area directly surrounding the contaminated source, either to withdraw contaminated water for treatment or to inject uncontaminated water as a barrier. In the former method, the contaminated groundwater can be isolated by drilling a number of wells across the hydraulic gradient at positions down-gradient from the contamination source; the wells, which should extend to the bottom of the aquifer, can then be used for withdrawal of the contaminated groundwater for treatment and for eventual disposal of the water. (Alternatively, the treated water can be injected into another aquifer or other waterbody.) In the latter method, injection of uncontaminated water into wells drilled across the aquifer gradient can stop or reverse localized groundwater flow. Both withdrawal and injection of contaminated water are considered to be short-term measures.

The principle of withdrawal at down-gradient wells is illustrated in Figure F1. The removal of water at the first row of wells, "A", causes a trough in the watertable surface, so that a reversed gradient is produced slightly downstream from the wells. This reversed flow region is recharged from the sides. There will, however, be some volume of contaminated water which reaches the region below the wells. Diffusion, eddies, and recharging will cause a mixing of the contaminated water in this volume with uncontaminated water downstream since the densities of the two waters are approximately equal. A second row of wells ("B") can be drilled down-gradient to repeat the process and remove the contaminated water not removed by pumping in row A. There will again be a volume of slightly contaminated water which eventually reaches the region downstream from the second row of wells. This water, in turn, will mix to some extent with uncontaminated water. Withdrawal of water from several rows of down-gradient wells can thus reduce contamination in the aquifer to very low levels, but can probably never eliminate the contamination completely.

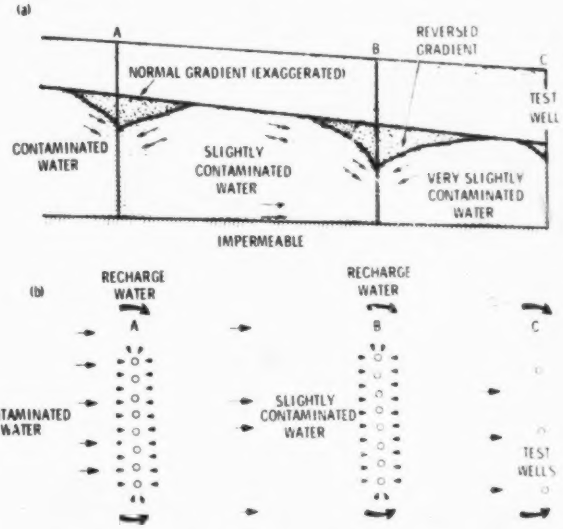

Figure F1. Pumping Wells Downgradient From the Source of Contamination

The number of wells required depends on the permeability of the substratum, the original hydraulic gradient of the region, the width of the band of contaminated water, the rate of discharge from each well, and the degree of cleanup required. Test wells ("C") should be drilled downstream from the second row to test whether the desired degree of decontamination has actually been met. The test wells themselves could be utilized to provide additional decontamination.

Withdrawal of contaminated water from an aquifer reduces the level of radioactivity in the down-gradient portions of the aquifer. However, it generates a very large volume of contaminated water above the surface. Consequently, withdrawal must be combined with an extensive storage and treatment program for all the withdrawn water.

Protecting the down-gradient portion of the aquifer by injecting water into a row of wells might also be thought to be attractive. The method has been used previously to impede the intrusion of salt water.[5] Whereas out-pumping removes the contaminant from the aquifer so that it can be safely disposed of by storage, concentration, evaporation, and so forth, in-pumping theoretically holds the contaminant in the aquifer indefinitely by forming a barrier to normal flow. If the aquifer does not have good lateral confinement, the contaminated water simply escapes around the edge of the pumping field, and a new field must be established down-gradient. From the new field, a second escape takes place, and still other fields must be drilled, ad infinitum, as shown in Figure F2. This process can take place with essentially no dilution of the contaminated water.

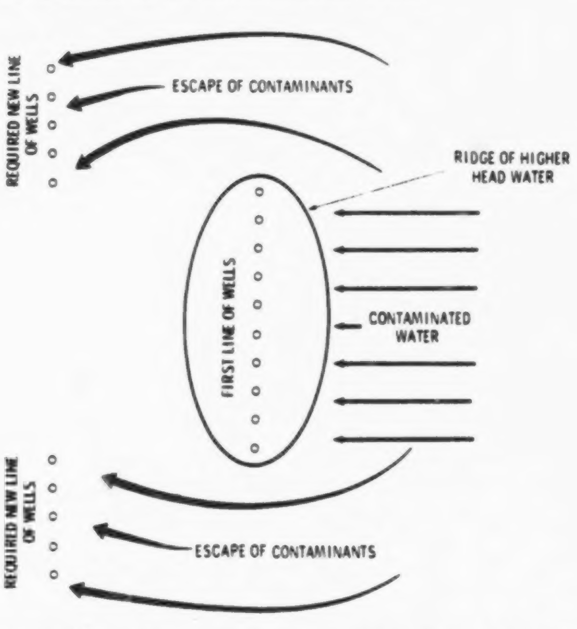

Figure F2. Installation of a Recharging Field in an Aquifer Without Lateral Confinement

If the aquifer is well confined laterally, the contaminated water is imprisoned in the aquifer as long as the barrier remains in place. However, if injection stops for significant periods of time, for example because of massive equipment breakdowns or seismic disturbances, the contaminated water continues on its way undeterred except for decay losses. As a result, injection should probably not be considered for radioactive contamination except as a stopgap measure.

The actual time it takes to drill wells for either withdrawal or injection depends on several factors, such as the depth of the wells, the number of wells to be drilled, whether or not the subsurface geology of the area surrounding the reactor is known (in many of the older reactors it is not), and, if the geology is known, the nature of the rock, and the number of qualified people available. One of the more important factors is the availability of rigs and equipment, especially on short notice. If time were critical, then more equipment and crews would be necessary to get the job done quickly.

The time it takes to drill each hole depends both on the drilling rates and on the setup and the takedown times for the drilling rigs. The average time including setup and takedown per several hundred foot hole might range from two to three days.*[6] Obviously, more time might be required in adverse conditions and/or for deeper holes.

F2.1.2 Lowering of Watertable

For a delayed source of contamination, such as leaching of the melt debris, pumping can be used to lower the watertable in the immediate area of the reactor. Leaching will then cease as the watertable drops below the debris. The contaminated water that can be obtained from some of the pumping can be stored temporarily and treated to remove radioactive materials. Figure F3 illustrates how the watertable is lowered at a single well, with the drawdown curve ("watertable after pumping started") representing the distance the watertable drops.[7]

A series of wells around the accident site could be pumped to lower the watertable from the entire area surrounding the melt debris. After pumping ceased, the water level would gradually rise or recharge to its original level, so that pumping is considered as a short-term solution to be used in conjunction with other interdiction methods, such as grouting.

Factors to be considered in determining the feasibility of lowering the watertable include the location of the melt debris, the soil conditions, especially the permeability, the distance to any nearby surface body of water, and the depth to a continuous impervious stratum. Pumping times depend on the quantity of water pumped and availability of equipment and crew.[8]

In investigating the practicality of lowering the watertable in the vicinity of the melt, the amount of drawdown which occurs in any given substratum for a specific amount of pumping is important. Equations for predicting the amount of drawdown in both confined and unconfined aquifers are given in Addendum F1 of this appendix. In particular, the drawdown resulting from pumping at wells evenly spaced on a circle centered on the reactor is considered in that addendum.

After the pumping rates are computed for various numbers of wells chosen so as to give the desired drawdown at the location of the melt debris, one can choose a combination based on cost and considerations such as whether the amount of water to be handled is reasonable. Some combinations of permeability, aquifer thickness, and hydraulic gradient may totally preclude the possibility of achieving the necessary drawdown. Even in cases where drawdown is possible, it must be considered at best to be a temporary measure.

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*The national average drilling rates for oil and natural gas are probably representative and are approximately 50 meters/hr (1200m/day) in soft rock, 15m/hr (360m/day) in medium hard rock, and 5m/hr (120m/day) in hard rock.[6] The daily average footage for drilling in all types of rock includes the time required for setup and takedown of the rigs and is approximately 312 ft/day or 95m/day. However, these petroleum industry rates are generally for relatively long holes (more than a 1000 meters) for which the setup and takedown times are a minimal contribution to the total time.

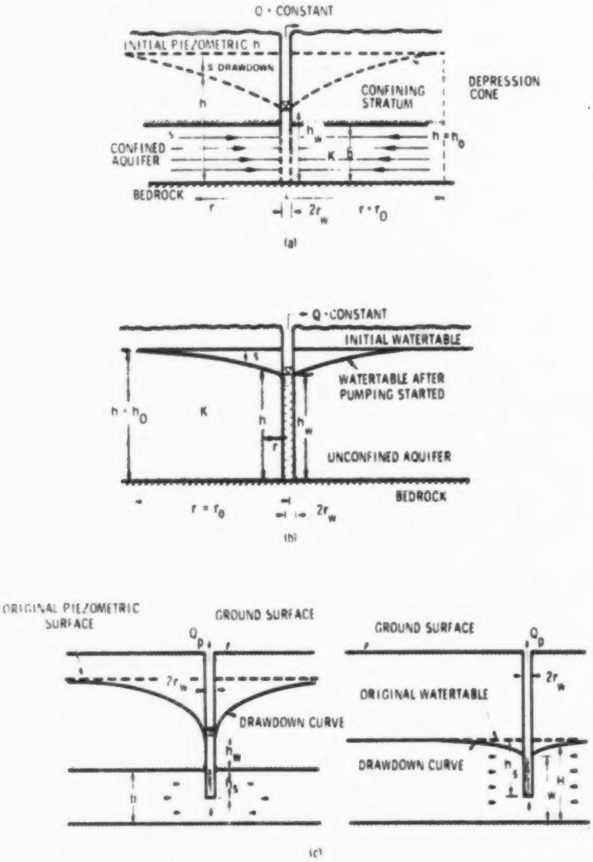

Figure F3. a) Radial Flow to a Well Completely Penetrating a Confined Aquifer

b) Radial Flow to a Well Completely Penetrating an Unconfined Aquifer

c) Radial Flow to an Incompletely Penetrating Well in a Confined (left) and Unconfined (right) Aquifer

Mixing of sumpwater with the groundwater could not be avoided by this method. If sumpwater release occurred, then all the water pumped during, and for some time after, that release would need to be treated extensively. If the water containing the sumpwater constituents were not treated, then pumping to lower the watertable would drastically hasten the entry of the sumpwater contamination into the environment. Likewise, the water which was initially pumped from around the melt debris would need to be treated. This would require the construction of rather extensive storage and treatment facilities at the site.

The advantages of drawdown pumping,[2] when it is possible at all, are that significant groundwater contact with the radioactive material may be delayed, thus allowing time for a more permanent solution (e.g., grouting) to be carried out. Drawdown pumping also gives additional time for decay to reduce the radioactive content of the material.

As a numerical example, assume that an unconfined aquifer with a water depth of 150 feet underlies the reactor. The permeability is assumed to be 100 gpd (gallons per day)/ft^2 (not untypical of sand) and the hydraulic gradient of the aquifer is taken to be 1/1000. If it is desired to draw the watertable down an additional 50 ft under the reactor, the necessary pumping rates (in gallons per minute) and the resultant height of water at each well after drawdown are as given in Table F2 for 2-5 evenly spaced pumps, each located 200 feet from the reactor center.

Table F2. Necessary Pumping Rates for Illustrative Example

Number of Wells	Rate Each (gpm)	Water Height at Each Well (ft)	Total Pump Rate (gpm)
1		Not possible	
2	400	39.6	800
3	288	67.3	864
4	242	77.3	968
5	210	81.3	1050

It might be decided that 400 gpm (gallons per minute) would be impractical for a well with only 40 feet of penetration into the waterbearing soil, so that the use of only two wells would be ruled out. Note, however, that little would be gained in going from 3 to 4 wells, except for a margin of error. Accordingly, it might be decided that three wells would be adequate. The amount of water to be disposed of (approximately 10^6 gallons per day) should present no problems if contamination were not significant.

This example is intended only to illustrate one possible set of circumstances. Under some conditions, a single well might be adequate for dewatering. Under more adverse conditions, dewatering might be completely impossible, or else might require a prohibitively large number of wells.

F2.1.3 Installation of Grout Curtain

A waterproof barrier might be formed by surrounding the melt debris with grouting. This would involve the injection of certain materials into the adjacent rock or soil to seal all the voids, cracks, and seams. Such a barrier could isolate the radioactivity from the accessible positions of the hydrosphere.

In general, grouting would be one of the more involved procedures which might be implemented for interdiction close to the radioactive source. Installation of a grout curtain would require a series of procedures. If the geology and hydrology of the area were not known beforehand, time would have to be spent taking core samples and otherwise examining the substrata. Site conditions, such as the nature of the formations and their fracture permeabilities, would have to be studied in advance to determine what materials, additives, and pressures should be used in the grouting.[*,9]

After the subsurface region were adequately characterized, grouting procedures could commence. Prior to grout injection, walls of openings would have been washed out so that the grout would adhere. In fractured rock, a dilute silicate might be injected beforehand to form a lubricating silica gel film.

Knowledge of subsurface conditions would be used to choose the appropriate type of grout. Two materials commonly used for grouting are Portland cement slurries and chemical grouts. Portland cement slurries, which are more popular due to their relatively low cost, are routinely used when the sizes of the voids and fissures are large enough to allow the slurry to be forced under pressure into the desired locations. Chemical grouts are used when smaller openings and crack widths must be penetrated. Chemical grout mixtures have a wide range in gel times and strengths and can be used to seal off an area even in flowing water conditions.[10]

For rocks and sediments having uniform porosities, cement grouting can be effective for thousands of years. Chemical grouting may be good for equally long times, but not enough information is available yet to establish such a long time range; it is known that chemical grouting can be good for at least 100 years.

Injecting grout into rocks presents problems since cracks are not generally continuous in a horizontal plane and they are not of uniform size. Also, the cracks may not be of sufficient width for the grouting to get through. If cracks are 1/16 in. or greater in width, some type of grouting can be injected. However, if the cracks are less than 1/16 in. in width, it may not be possible to inject grouting without further preparation.

Artificial fracturing of the rock surrounding the drill holes is possible, and is widely practiced in the petroleum industry. Fluids can be injected at high pressure into cracks to lengthen and enlarge them. Sideward holes can be drilled from the main drill hole, with further artificial fracturing.

All of these fracturing techniques necessarily increase the cost and the time required for grouting. In addition, in grouting around any melt debris, such fracturing techniques could also increase the possibility of early release of radioactivity to accessible portions of the hydrosphere. In some locations, the increased possibility of early release due to fracturing might be thought to even outweigh the potential advantages of grouting.

One of the important factors affecting the time required to install a grout curtain is the availability of drilling and grouting equipment and trained operators. Special equipment may also be needed for handling radioactive materials. In general, all the required equipment and personnel would need to be transported to the site.

An example of one possible grouting configuration would be that described in a previous report.[1] The hypothetical debris source is 21 m in diameter and 16 m high, and is located directly below the containment structure. Grouting would be pumped down through slant-drilled holes extending below the core debris on a 190 m (623 ft) circumference (or diameter of 60.5 m) to form a waterproof barrier surrounding the debris source (Figure F4).[1]

However, this configuration ignores the environment of the reactor. The buildings surrounding the containment area need to be considered in discussing the feasibility of grouting. Figure F5 shows diagrams of typical BWR and PWR two-unit nuclear stations.[11,12]

The area of a reactor building in either diagram corresponds to the approximate maximum grouting dimensions in the example discussed above. It would seem virtually impossible to consider grouting in such a confined area when each reactor unit is surrounded by a network of pipes and a variety of buildings. The structures surrounding the containment could perhaps be leveled so that grouting could

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[*] Initial procedures to determine the geology of the area would include:

1. ascertaining the degree of continuity of joint since discontinuity promotes strength (continuous joints represent zones of weakness usually failing at lower stress values than the rock material itself);
2. investigating amounts and direction of dip of jointing and degree of stratification;
3. investigating the nature of the joint surfaces since these dictate resistance to movement and influence the distribution of stresses (the actual thicknesses of joint zones may have a bearing on groutability and these can be affected by several factors, such as weathering and the movements of groundwater);
4. analyzing actual blocks of rock material to determine their physical, mechanical, and dynamic properties; and
5. investigating subsurface water conditions.

be done just around the reactor building. However, attempts to remove the attached buildings could result in damage to the containment building itself and could cause the release of radioactive water from the auxiliary buildings. Furthermore, the leveling of the attached buildings would necessitate the disposal of all the contaminated materials produced by the razing. As a result, the grouting would probably have to surround at least the entire nuclear unit.

One possibility would be that the holes could be slant drilled at different angles from the perimeter of a rectangle which could avoid all the buildings. For any of the units depicted in Figure F5, a possible configuration for grouting might be a rectangle approximately 500 ft x 800 ft. The perimeter would be about 790 m (2600 ft) or slightly greater.

Grouting could be generally assured if the holes had a separation of no more than 1.5 m (5 ft) at the surface.[9] It should be noted that the separations would be much less under the reactor because the holes would become crowded together at the apex. If the holes were spaced on the 500 ft x 800 ft rectangle, about 520 holes would be required. If the holes were each 110 m (361 ft) long, the apex would be 78 m (260 ft) below the ground at the reactor center.

From the drilling rates referred to in Section F2.1.1, approximately 3 days might be required to drill a 110 m hole. However, the required slanted holes would be more difficult to drill. There would also be a rather difficult surveying job involved in assuring that the holes stayed on course with considerable precision. Accordingly, it is estimated that 6 to 8 days would be more reasonable for a slanted hole. To ensure adequate grouting, core samples would probably need to be taken at least at 15 m (50 ft) intervals. As a result, the initial holes could take considerably longer than 6 to 8 days to complete. If 20 drill rigs were used, drilling of the entire 520 holes for the rectangle could be completed in 7 to 9 months, in favorable conditions.

Grouting is usually done on a splitting pattern; that is, the initial holes are evenly but widely spaced. After these are filled, intermediate holes are drilled and grouted. In principle, the pattern is repeated until grouting is assured. However, splitting-pattern grouting along a single row such as that just considered is often not adequate to ensure a complete cutoff.[13] Instead, a multiple-row grouting approach is required. In such cases, a three-row pattern is usually adequate (Figure F6). First, the two outer primary rows are drilled and grouted, each on a splitting pattern. Then the inner secondary row is drilled and grouted on a repeated splitting pattern until cutoff is assured.[13]

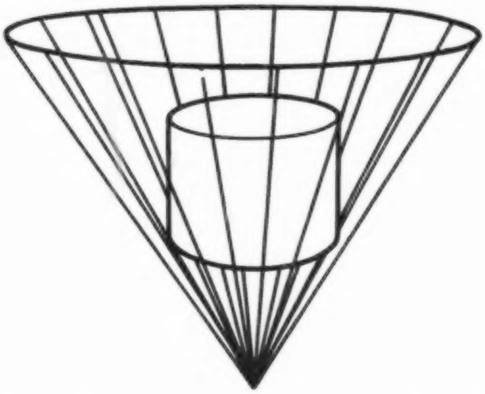

Figure F4. Possible Grouting Configuration for the Melt

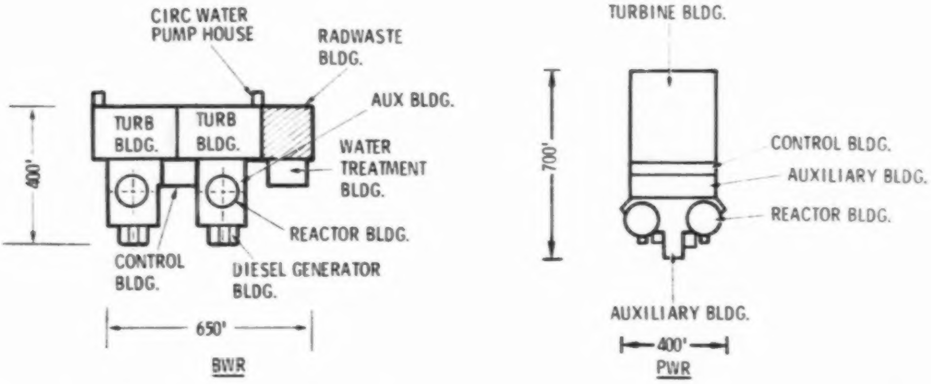

Figure F5. Typical BWR and PWR Nuclear Station

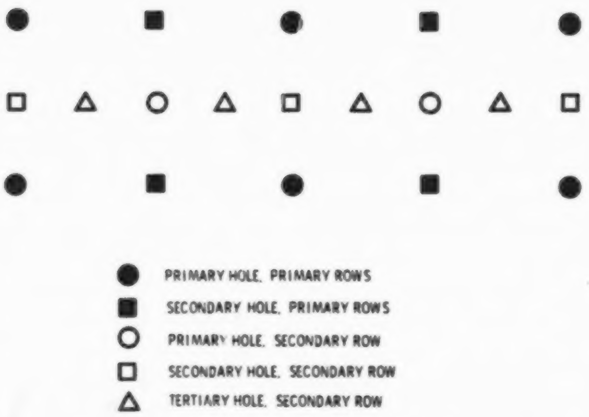
● PRIMARY HOLE, PRIMARY ROWS
■ SECONDARY HOLE, PRIMARY ROWS
○ PRIMARY HOLE, SECONDARY ROW
□ SECONDARY HOLE, SECONDARY ROW
△ TERTIARY HOLE, SECONDARY ROW

Figure F6. Three-Row Grouting Pattern With Two Primary Rows

Such a multiple-row approach could drastically increase the total time required to complete the grouting. It might be possible to use more drill rigs, reducing the time required. However, it would be necessary to maintain a minimum separation between rigs to avoid interference. Also, scheduling would be more difficult as more rigs were used. For these reasons, it is believed that even with exceptional effort, the drilling portion of the task often could not be completed in much less than one year. However, effective temporary cutoff of the debris might be obtained much earlier than this, after completion of just the first row.

It has been assumed that no major difficulties are encountered. Very hard rock strata, or requirements for extensive fracturing, or uneven layers of varying hardnesses could easily double the time required to complete the grouting.

The configuration just considered for grouting is only one possibility. If the rock strata beneath the site were not horizontally layered, then the angles at which the slant drilling were done might be different on various sides of the configuration. If a sufficiently thick impermeable layer of rock existed not too far beneath the melt debris, then grouting might be done vertically just to that layer; the impermeable rock would then serve as part of the barrier. For a single isolated unit, it might be possible to grout around a smaller area. Different configurations could require much different times to complete than those just estimated.

Site preparation would have to precede any drilling. The necessary site preparation — concrete pads for at least some of the holes, surveys, decontamination, and so forth — before drilling started is estimated to require approximately three months.

After grouting around the sides and the bottom of the melt debris, it might be necessary to seal off the top of the debris. To accomplish this, additional grouting might be done either under or at the surface of the site so that the debris would be completely encased in grout. Alternatively, an above-ground structure enclosing the containment and any other necessary buildings might be built. Additional time would be required to complete this work inasmuch as such work could probably not be initiated until the completion of all the grouting under and around the debris. The time required to seal off the top of the debris could range from days to years, depending upon the approach employed.

The total time required for grouting and all the related procedures is thus estimated to range from 15 months to 3 years. Under favorable conditions, the task might take considerably less than this estimated range. Under exceptionally awkward conditions, the task might take longer. Additional time would also probably be required if the geology and the hydrology of the site were not well known in advance.

It should be pointed out that grouting would nearly always be possible, provided that sufficient time were available. If the original work proved not to give adequate protection, additional holes could be drilled and grouted. However, if the task consumed too much time, some release (particularly sumpwater release) might occur before the job were finished at certain sites.

F2.1.4 Other Methods

If a narrow, laterally confined aquifer with a well-defined unidirectional hydraulic gradient were the source of the groundwater at the site, a grout curtain could be installed either upstream or downstream from the reactor. Such a barrier would be worthwhile only if the bottom of the aquifer could be completely sealed off; otherwise, water would simply leak under the curtain. This type of barrier would preclude downstream use, so that alternate sources of water would be required. The barrier could not be considered permanent, and long-term monitoring for leakage would be required. An upstream barrier would be better than a downstream barrier.

Some sumpwater releases might be precluded by pumping out the containment. In this method, a hole could be drilled at a shallow angle to penetrate the containment sump. The hole would have to be completely cased and well grouted all the way to the containment wall. Most of the sumpwater could then be pumped out for temporary storage in impermeable tanks. Alternatively, piping connections to the sump could be used to pump out the contaminated water.

It has been suggested that a slurry trench could be dug around the melt debris. This would require the excavation of a rather deep trench all the way around the containment. This trench would then be filled with slurry. Extensive pumping, along with treatment of any contaminated water pumped, would need to accompany the entire process. Although the barrier could generally be assured as far as it went, such a trench would not necessarily make a better barrier than a more complete grout curtain. In fact, as time permitted, construction of a more complete barrier would often be indicated.

F2.2 Source Interdiction as a Function of Site Features

Effective isolation of the radioactivity released into the liquid pathways after a meltdown accident depends to a large extent on the characteristics of the reactor site, as well as on the types of releases which occur. To be worthwhile, source interdiction must be completed before significant quantities of the radioactivity have been carried away from the vicinity of the accident. The time before which interdiction must be carried out, and hence the feasibility of successful interdiction, depend strongly on the hydrology of the site. The feasibility also depends upon the presence of certain engineered features such as underdrain systems.

Constructed features at the site such as underdrains or dewatering systems can drastically hasten the entrance of sumpwater either into the ground water or into nearby surface waterbodies. Temporary underdrains are often installed during construction of reactor containments to remove water and are then normally closed off upon completion of the project. Permanent underdrains, which are not sealed off following construction, are used after construction to lessen subsurface water loads. The permanent systems are economical and provide greater plant operational dependability for the following reasons:[14]

1. they reduce structural requirements such as wall thickness in earthquake zones;
2. they keep the water level below that of any liquifiable soils;
3. they prevent seepage water from entering the lower building levels; and
4. they minimize costs by reducing the amount of interior drain water to be treated.

There are two types of underdrain systems: gravity flow (passive) and pumped flow (active). Each generally consists of a collector system, a drain system, and a discharge system. In one possibly typical gravity underdrain, water collected by a series of vertical and horizontal pipes flows to a nearby drainage ditch. The flow of water along the entire route of the system is by gravity. In one typical pumped system, pumps are used to lower the watertable from a gravel fill beneath the containment. The water flows through vertical and then horizontal pipes to a nearby sump cavity, from which it is finally pumped to a holding pond. The majority of underdrains are contained alongside and under structures, with some systems having pumping systems within larger systems, such as reactor or auxiliary buildings, to discharge collected water.

For the liquid pathways, the gravity flow system is probably more of a threat than a pumped flow system. A release of sumpwater into a functioning gravity flow system might result in its almost immediate entry into the environment, making source interdiction extremely difficult or even impossible.

In contrast, the functioning of a pumped flow system might be advantageous to interdiction. If the pumped system remained intact in the event of a meltdown, and if the melt did not progress too deeply into the soil, the watertable might remain sufficiently depressed that groundwater did not contact the melt. The release of radioactivity by leaching could then be delayed as long as the system functioned appropriately. However, failure of a pumped system could result in recharging of the watertable to its original level. This could cool the melt and initiate leaching. In addition to being used to lower the watertable, a properly-functioning pumped flow system could possibly be utilized to gather any escaping sumpwater. Thus, its release to the environment could be circumvented.

It is not generally known whether permanent underdrains exist at many of the older reactor sites. There are thought to be at least a dozen sites with such permanent underdrain systems. At such sites, the probability of early sumpwater release could be significantly larger than it is for sites without permanent underdrain systems.

Constructed features other than underdrain systems might also affect releases to the hydrosphere. For example, during construction the region beneath and around the containment basemat might be altered such that a direct path were created between the region and the ground surface. Thus, the rates of escape of sumpwater and depressurized containment gases to accessible portions of the environment might be significantly increased. At such sites, successful source interdiction of the radioactivity in those releases might be precluded.

The hydrology of the site is also very important in determining the feasibility of most source interdictive procedures. Withdrawal of contaminated water would only be practical in areas of relatively low groundwater flow rates; in other areas, the amount of water to be stored and/or treated could be too large to handle. A relatively low natural watertable at the site would tend to make withdrawal a more useful procedure; a high watertable would increase the amount of water to be pumped and treated, unless pumping of the upper uncontaminated water were done separately. Injection of water as a barrier would be feasible only in regions in which water of sufficient quality was readily available.

Soil conditions and rock types also can affect mitigating actions. If rock fractures are too small, then artificial fracturing must be utilized to ensure the feasibility of grouting. Nonuniform permeabilities can make both grouting and pumping difficult and even completely impracticable.

The hydrological conditions under the reactor might be such that extensive source interdictive procedures would not even be required. For example, the watertable might lie far beneath the lowest depth of penetration of the melt. If depressurization and sumpwater releases did not occur, then release of radioactivity to the environment would be extremely slow. If precipitation were prevented from percolating down to the melt, then release would be by diffusion only. Such a location might be regarded as being a relatively safe repository for the melt debris.

Another condition which might preclude the need for interdictive procedures, other than monitoring, would be entrapment of the melt in a thick and rather impervious layer of material, such as clay, beneath the reactor site. If the intrusion of the melt into the material did not degrade the layer's integrity with respect to permeability (and if integrity had not been previously disturbed by construction activities), then the impervious layer might form a comparatively secure location for the melt debris.

At its best, source interdiction, as it has been considered here, is a long-time, but not necessarily a permanent, solution. The cost estimates to be presented in Section F2.4 for the previously considered procedures do not take this into account, and therefore the total costs could ultimately be much higher than those considered in this appendix. For example, in the case of grouting, an initial attempt might not result in a continuous barrier. Furthermore, the grouting could fail eventually by deterioration of the grout itself or as a result of seismic activity. In time, the melt and its grout curtain might be resealed inside another grout barrier. Alternatively, the melt debris might be moved to a permanent waste repository. The problems encountered in implementing such permanent solutions are not considered here.

For many of the older facilities, knowledge of subsurface geology may be limited, thus making it difficult to determine the necessities and the feasibilities of many source interdictive measures in an accident situation. This lack of knowledge could hamper the reasoned consideration of possible approaches to source interdiction at some sites. Furthermore, it could lower the probability of success of any actions which were taken.

F2.3 Source Interdiction as a Function of Near-Field Contamination

The previous subsections have assumed that ground contamination at the accident site would be minimal. However, that would not necessarily be the situation. In the event of a meltdown accident, some form of atmospheric release would inevitably accompany the hydrospheric release. Some of that material released into the atmosphere could contaminate the ground around the site. The most probable atmospheric releases would be such that even near the reactor there would be a possibility of taking mitigating actions at a relatively early time. However, the probability of a more severe release is far from negligible, and therefore it is necessary to consider the possibility that near-field contamination could preclude some immediate interdictive responses close to the site.

F2.3.1 Near-Field Doses and Dose Rates

The dose received by an individual in the vicinity of the reactor can be estimated roughly by using the Reactor Safety Study Consequence Model.[15-17] Details of the calculations performed for this study with that model to estimate the near-field dose rates are presented in Addendum F2. The resulting dose rates are displayed in Figure F7 and are used here to consider the effects of near-field contamination on source interdictive procedures.

F2.3.2 Effect of Near-Field Contamination on Implementation of Source Interdiction Procedures

The workers implementing any clean-up procedures for the accident site would be subject to the limits for occupational exposure in emergency situations. Absolute limits for such exposure currently do not exist. The recommended limits are discussed in Addendum F3.[18-22]

If it were decided that a 25 rem dose in a single working day were acceptable in view of the expected benefits, the limiting dose rate would be 75 rem/day.

(Note that each worker could be employed only one day on the project at this rate.) According to Figure F7, there would be approximately a 60% chance that the dose rate would be less than this immediately. The probability would become greater than 0.9 after one month and nearly unity after 1 year.

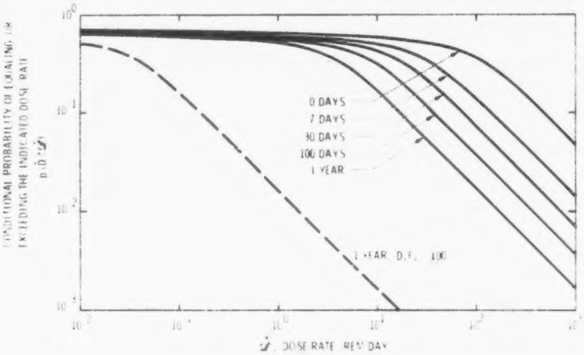

Figure F7. Complementary Cumulative Distribution Functions for Dose Rate at 402 m From the Reactor With No Shielding Included and With Resuspension Ignored.

Such high dose rates would probably be unacceptable for long-term operations. However, some selected workers might be employed for many days at rates of 0.1 rem/day. At this level, there is about a 40% chance at any time up to one year. With decontamination, the probability goes up to 0.83. Protective clothing for the workers could lower the effective dose rates in all cases, and therefore could increase the length of time a given worker could be employed in contaminated areas.

As a result, on the basis of the calculated dose rates presented in Addendum F2, it can be seen that even if access to the significantly contaminated areas were required, careful decontamination and protective equipment could make most operations possible. In nearly every case, some means could be found to circumvent the limitations imposed by contamination. The principal results of the near-field contamination would be to increase the total cost and effort required for interdiction and to delay the initiation of some interdictive procedures.

In addition to the contamination produced by an atmospheric release, local radiation from within the plant (from the contained source) could be high and therefore hinder some interdictive procedures. For example, intense radiation fields could exist in the vicinity of the recirculation piping and heat exchangers, or near spills of contaminated water. If the containment ruptured at a low enough level, then both the melt itself and radioactive materials deposited out within the containment could constitute potential radiation hazards. Interdiction within the reactor complex would require careful consideration of such local effects.

Another potential source of local radiation would be due to the interdictive efforts themselves. For example, if any drilling were carried out in the vicinity of the core, the drilling mud and tools could become seriously contaminated. The contaminated materials would have to be disposed of in such a way that further interdictive efforts were not hampered.

The most important practical effects of near-field contamination on interdictive efforts would be the requirements for monitoring of the on-site radiation, for using protective clothing, for restricting working techniques, and for making the difficult decisions as to the exposure limits to set for the workers. The methods used for estimation of the dose rate are quite crude. However, it should be noted that the most important conclusion is that contamination near the reactor usually would not be a serious barrier to most interdictive efforts. Even if the calculated dose rates were an order of magnitude higher, this conclusion would not be significantly different.

F2.4 Costs

Approximate costs for source interdictive measures are discussed in the following subsections and in Addendum F4. These costs would vary from region to region, and would be influenced by the following:

1. the scale of monitoring necessary;
2. the amount of radiological controls indicated; and
3. the dimensions of the required actions.

F2.4.1 Direct Costs

The direct costs associated with source interdiction include both those incurred for monitoring and those resulting from implementation, such as construction costs. Monitoring is discussed briefly in general terms in the following subsection and in the subsection on monitoring for pathway interdiction, F3.4.1.1. An example of specific monitoring costs is presented in Addendum F5. Factors affecting the costs of the implementation of the measures mentioned in Subsection F2.1 are reviewed below. Examples of such implementation costs are detailed in Addendum F4.

The examples of costs considered in the addendum on source interdiction costs (Addendum F4) and in the addendum on pathway interdiction costs (Addendum F5) are merely meant to be illustrative and are not necessarily representative of the interdiction costs which would actually be incurred. Furthermore, the indicated ranges of costs are not intended to be all inclusive.

F2.4.1.1 Monitoring

Monitoring in the immediate vicinity of the accident site would be a required concomitant of any interdictive procedures. Monitoring would be needed to ensure that the interdictive actions which were taken adequately protected the environment. For any type of source interdiction, very thorough monitoring of the immediate accident vicinity would be required for many years after the meltthrough.

For the methods just considered for source interdiction, monitoring for possible groundwater contamination would be most important. A significant part of the monitoring cost would be that of the installation of wells.

F2.4.1.2 Wells

Wells would need to be installed for almost any type of source interdictive effort. At a minimum, they would be needed to check on the groundwater in the vicinity.

Some of the factors affecting the costs of installing wells include the required depths, the prevailing substrate conditions, such as the hardness of the rock, and the necessary precision. Obviously, any type of adverse condition which tended to increase the drilling difficulty and/or time would tend to increase the corresponding well costs.

Pumping costs would depend primarily on the rate of pumping needed and the total amount of water considered. The larger either of these two factors were, the greater would be the associated costs.[23-25] The costs of treatment of contaminated pumped water would include those for installation of appropriate treatment and storage facilities plus those for subsequent processing. These costs would generally be expected to increase with the level of contamination.[26]

F2.4.1.3 Grouting

The costs of grouting would depend upon the size of the region grouted and the subsurface conditions encountered.[27] Any factors which increased the complexity and/or the magnitude of the grouting operation would tend to increase the cost. The primary cost of the grouting would usually be that due to drilling the indicated wells.

F2.4.2 Indirect Costs

Indirect costs of source interdictive reflect the social impacts of interdiction. These costs are thought to be minor for source interdiction since interdictive measures would be based at the plant, and contamination of aquifers and surface waterbodies would be minimal.

F3 Pathway Interdiction

Until now, the discussion in this appendix has focused on interdictive procedures which could be taken in the immediate vicinity of the accident site. However, in situations with relatively rapid releases to accessible portions of the hydrosphere, such actions might not be possible. Furthermore, certain siting considerations might render such source interdictive procedures infeasible. In these circumstances, the only mitigating procedures which would be possible would be pathway interdictive ones. That is, once the radioactivity had reached the portions of the environment accessible to humans, only direct intervention at some point along the exposure pathways could lessen the radiation dose to the population. Pathway interdictive procedures might also be required if source interdictive procedures either were not executed for some reason or else were not successful in containing the radioactivity.

F3.1 Methods of Pathway Interdiction

For any reactor located at some distance from all surface waterbodies, human exposure, if it occurred, would generally be via well usage. For reactors located relatively close to surface waterbodies, exposure would usually be via contaminated surface water usage. The specific pathway interdictive procedures employed at a given site would depend on both the characteristics of the affected waterbody(ies) and the sizes of the potentially exposed populations.

All of the procedures for pathway interdiction can be divided into two types: those involving interruption of the flow of radioactivity along some or all of the pathways to the human population and those involving removal of radioactivity from the entire environment. The former interruptive procedures include confiscation of polluted food, substitution of drinking water, and denial of access to polluted areas; the latter restorative procedures include treatment of polluted water in a waterbody, dredging of contaminated sediment and decontamination of land. Monitoring is an integral part of all pathway interdiction plans.

F3.2 Pathway Interdiction as a Function of Site Features

F3.2.1 Surface Features

The characteristics of the contaminated waterbodies which determine the effectiveness of pathway interdictive procedures at a specific site include type(s) of water, flushing times, and sedimentation properties. Obviously, at certain sites, some pathways are not necessary because of the type of water contaminated (Table F4). For example, at saltwater sites, drinking water and irrigation pathways are not encountered and therefore interdictive procedures for these pathways are not required.

Table F4. Pathways Potentially Requiring Interdiction

	Drinking Water	Aquatic Food	Irrigated Food	Shoreline Usage	Immersion
Aquifer	x	-	x	-	x
Lake	x	x	x	x	x
Estuary	-*	x	-*	x	x
Ocean	-	x	-	x	x
River	x	x	x	x	x

*May be a pathway in the upper portion of a salt-wedge estuary.

The feasibility of pathway interdiction at any site is a function not only of the dominant pathways affected but also of the scale of the overall effort required. The scale of the effort depends upon the rate of spread of the contamination in the environment. In some waterbodies, the initial rate of dispersal would be so large that indicated pathway interdictive procedures would be too extensive to be practical.* In contrast, in other waterbodies, the initial rate of dispersal would be small enough that some types of pathway interdiction would be reasonable.

In general, the magnitude of the required effort would be much less for interruptive procedures at any given site than for restorative ones there. Therefore, the former can be regarded as being more practical than the latter.

The relevant hydrological features of each site which would affect the need for interdiction for each exposure pathway are considered in Chapter 3 and are summarized in Table F5. The factors associated with the pathways themselves which are pertinent to interdiction are discussed in Chapter 4 and in Appendix E. The resulting implications for pathway interdiction are reviewed briefly in the following several subsections.

F3.2.1.1 Dry Sites

When there are no nearby bodies of surface water present, the only pathways potentially requiring interdiction are drinking water and irrigation. At such sites, exposure could occur both from wells penetrating contaminated aquifers and from springs issuing from these aquifers. In general, pathway interdiction would consist of denying use of water from such aquifers. Leakage from springs would generally require more extensive actions. Except perhaps where such leakage was significant, pathway interdiction at dry sites would usually be very easy and effective.

F3.2.1.2 Great Lakes Sites

All of the five exposure pathways considered in this report are present at any Great Lake site. The relative importance of the various pathways, and hence the relative need for interdiction, would be determined, in large part, by the characteristics of the lake itself (Table F5).

Inasmuch as substantial buildup of activity in the deposited sediment generally would occur only over a relatively long period of time, dredging of sediment would probably not be required, except perhaps in the immediate vicinity of the accident. Shoreline cleanup would also not be necessary other than near the site due to the low net deposition rate of radioactivity onto the shoreline.

More important pathway interdictive procedures at a Great Lake site might include treatment of drinking water taken from the lake or else substitution of nonlake drinking water, along with condemnation of the finfish catches for the entire lake. Even after releases of relatively short duration, such actions might need to be taken for many years due to the very slow flushing times of the lakes.

The most important pathway interdictive procedures at a Great Lake site would usually be those relating to drinking water. However, due to the substantial rapid initial dilution of radioactive material entering the lake, the concentrations of radioactivity might be too low to warrant interdiction even for that pathway, except perhaps close to the site of the release.

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*Although pathway procedures are always theoretically possible, their potential implementation costs may obviate their use. In practice, if the dispersal is such that the resulting concentrations of radionuclides are below those dictated as being "important" by the appropriate exposure criteria (see Subsection F3.3 and Addendum F3), then pathway interdictive procedures are not thought to be warranted and therefore will probably not be carried out.

F3.2.1.3 Estuary Sites

Only the aquatic food, shoreline and immersion pathways would usually be present at an estuary site. The amount of radioactivity in each of these pathways would be very dependent upon the general characteristics of the estuary (Table F5).

Inasmuch as there can be a rapid, substantial deposition of contaminated sediment onto the floor of an estuary, dredging might be required. The areas to be dredged could range from small localized spots to rather extensive portions of the entire estuary. Estuary characteristics can vary greatly, so that it is not easily determined in general where zones of radioactive material might occur. Because of the tidally induced dispersion which occurs in an estuary, significant contamination could be found upstream as well as downstream from the accident site. Some type of shoreline cleanup would probably also be necessary due to the heavy sediment load often deposited during high water stages.

Other pathway interdictive procedures at an estuarine site might include condemnation of both the finfish and the shellfish catches for the estuary for significant periods of time both during and after the release. This is partly because contaminated organisms and sediment could form significant secondary sources of contamination for the ecosystem long after the radioactivity had been flushed from the water. The need to interdict the shellfish harvest would probably be greatest, due to both the relatively high concentration of some radionuclides in shellfish and the comparatively large size of the shellfish harvest for some estuaries.

The most important pathway interdictive procedures at an estuary would be those pertaining to aquatic food ingestion and to shoreline usage. The effectiveness of any pathway interdictive procedures would depend in part upon the rate of release of radioactivity into the estuary.

F3.2.1.4 Ocean Sites

The pathways of potential importance at an ocean site are the same as those at an estuary site. The relative importance of each of the pathways would be dictated by the hydrological characteristics of the region (Table F5).

Just as for Great Lake sites, shoreline cleanup would probably not be required for most nonestuarial ocean sites, except perhaps in the immediate vicinity of the accident, due to a negligible deposition rate of radioactive material onto the shoreline. Dredging would also not generally be indicated.

More necessary pathway interdictive procedures at an ocean site might include condemnation of the fish catches for the vicinity. Such actions would probably need to be taken only during the period of the release and for a short time afterward. The relatively rapid flushing present at most ocean sites would tend to quickly disperse the radioactivity to regions of lesser potential contact.

Table F5. Summary of Hydrological Properties

Waterbody	Dilution	Flushing	Relative Concentration in Water	Soil or Sediment Effects
Aquifer	Small	Very Slow	Large	Soil retards many radioclides significantly.
Lake				
Great Lake	Large	Slow	Small	Slow sedimentation rate; some retardation of nuclides; some burial.
Other Lake	Moderate	Moderate	Moderate	Moderate sedimentation rate; some retardation; some burial.
Estuary	Moderate	Moderate	Moderate	High sedimentation rate; potentially substantial retardation of radionuclide movement; potentially significant burial.
Ocean	Large	Fast	Small	Very low sedimentation rate.
River	Moderate	Fast	Moderate	Variable sedimentation rate; large movement of sediment in free stretches; reservoirs are sediment sinks.

The most important pathway interdictive procedures at an ocean site would be those involving the aquatic food pathway. However, the significant early dilution of radioactivity which occurs in open coastal regions would probably preclude the need for interdicting even the aquatic-food pathway, except perhaps in the vicinity of the release.

F3.2.1.5 River System Sites

All of the five exposure pathways considered in this study would exist at a site on a river system. The radioactivity in each of the pathways would depend upon the hydrological description of the river (Table F5).

Dredging of some downstream reservoirs close to the accident might be necessary. Shoreline cleanup for some reservoirs might also be required due to the potential of substantial sediment deposition by floods. In addition, floodplain cleanup might also be indicated in certain regions.

Other pathway interdictive procedures for a river system site might include treatment of drinking water taken from the river or else substitution of drinking water from other sources, condemnation of the fish catches and denial of water for irrigation purposes. Some of these interdictive procedures might be warranted for the complete length of the river system for the entire period of release and for some time after the release.

The most important pathway interdictive procedures at a site on a river system would be those relating to drinking water, aquatic foods and shoreline usage. Interdictive efforts would generally be expected to decrease in net effectiveness with distance downstream from the accident location.

F3.2.2 Subsurface Features

The hydrological conditions beneath the reactor ultimately determine the amount of radioactivity which reaches the exposure pathways. The hydrological conditions might be such that extensive interdictive procedures were not required (see Subsection F2.2). At such sites only monitoring would probably be necessary.

Natural hydrological conditions which would tend to permit the release of radioactive materials into accessible portions of the hydrosphere could sometimes be circumvented by interdictive procedures at the source, such as those already described in Section F2. If they could not be, then only pathway interdictive procedures, such as those just considered, could be utilized to mitigate the radiation exposure received by the human population.

F3.3 Pathway Interdiction Criteria

The commonly-used intervention criteria for non-occupational exposures are all based on the concentrations of the radionuclides in the relevant portions of the exposure pathways (see Addendum F3). For the liquid pathways, these concentrations depend, in turn, upon the concentrations in the water. Those concentrations depend upon both the rates of release of activity into the waterbody and the subsequent rates of removal from the waterbody, for example, by flushing.

It should be noted that the concentrations in a given waterbody would be more likely to exceed the MPC's (maximum permissible concentrations), or the concentrations which would result in certain projected doses, if the rate of release were relatively rapid and/or the rate of removal were slow. However, the total population dose is not necessarily related to these two factors. As a result, using MPC's or, equivalently, fixed individual doses as the interdiction criterion does not result in a uniform reduction in the population dose at different waterbodies, or even at a given waterbody. For example, the population dose at a given site is affected primarily by the total amount of radioactivity released (before decay) and essentially not by the concentration.*

F3.4 Costs

The costs of pathway interdiction can be either direct or indirect. Direct costs include both monetary outlays and losses and adverse social impacts. Examples are the following:

1. the outlay for pathway monitoring;
2. the outlay for interrupting a given pathway;
3. the value of the output of interrupted industries, for example, the value of fish from a "quarantined" fishery;
4. the outlay for providing an alternate source of supply, for example, the cost of providing an alternate supply of drinking water; and
5. the loss of jobs in severely impacted industries.

Indirect costs can also be either economic or social. Disruption of an industry in one sector of the economy can generate indirect losses in other sectors. Indirect social losses could be incurred because of the inconvenience and disruption caused by interdiction.

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*In contrast, the estimated health effects can depend upon the concentration.

F3.4.1 Direct Costs

Costs for the pathway interdictive measures previously discussed are considered in general terms in the following subsections. Specific examples of some pathway interdiction costs are presented in Addendum F5. All of these discussions of costs are not meant to be all-inclusive; rather, the intent is to be illustrative of the types of costs which can be incurred if various pathway monitoring and interdictive procedures are implemented.

F3.4.1.1 Environmental Monitoring

Monitoring of the exposure pathways would be necessary either if source interdictive measures were not taken or else if those actions taken were not completely effective. At a minimum, pathway monitoring would include measurement of the total radioactivity in any accessible water. If significant contamination were found in the water, then monitoring of portions of potentially affected pathways would also be indicated.

Costs would vary, depending on the nature of the release. The costs for monitoring would also be extremely dependent upon the local conditions present at the affected waterbody(ies). For example, if the waterbody into which the contaminated aquifer discharged were remote, did not have an outlet into potable water, and were not used as a water supply, the monitoring effort and related cost could be low. On the other hand, if the waterbody were located in a densely populated area, were extensively used, and discharged into other lakes and rivers, a more widespread effort would be required at much greater cost.

Unless adequate source interdictive procedures were employed, the pathway monitoring levels required would not decrease with time for many years. Monitoring could be required at significant levels for even decades after the accident.

F3.4.1.2 Pathway Interruption

The least costly type of pathway interdiction involves interruption of the flow of radioactivity at some point along the exposure pathway, usually close to the domain of contact with humans. In general, only the human population benefits from such actions; the rest of the populations in the environment do not. Interruptive procedures considered here include substitution and/or treatment of water supplies, confiscation and/or denial of finfish and shellfish, and relocation of certain portions of the population.

F3.4.1.2.1 Drinking Water

In general, some type of interdiction of the drinking water pathway would be one of the most important types of mitigating actions which could be undertaken at any freshwater site. The related costs would be very site dependent (see Addendum F5).

Costs for alternative water supplies include those for transportation, replacement of water, storage tank construction, and limited plumbing.[28-31] If additional wells are installed, drilling and piping costs are also incurred.

The amount of water transported to a community would vary according to a site location, as well as the contamination level of the water. At dry sites, where the sole source of water was an aquifer in which the water was highly contaminated, all water might have to be imported. For sites adjacent to surface waterbodies, it might be necessary to import water for drinking and cooking only.

As an alternative to substitution, treatment of the contaminated water might be considered. If there were existing treatment facilities, the only costs necessary might be those for improvement of existing facilities plus those incurred from handling and disposing of radioactive processing media.

Monetary costs for substitution and/or treatment of water supplies would not reflect the decreased standard of living which might occur in the affected areas. Furthermore, restricting use of water in industry could slow the growth and development of these areas until contamination was no longer at hazardous levels.

F3.4.1.2.2 Finfish and Shellfish

In general, the restriction (i.e., confiscation or denial) of aquatic-food harvests would be one of the more important types of indicated pathway interdictive procedures. If aquatic organisms in large areas were substantially contaminated, the costs of such procedures could be substantial (see Addendum F5).

The value of the finfish or shellfish in a fishery is a function of both the location and the type of waterbody involved. For commercial fishing, the direct loss could be measured by the sales value of the fish.[32-34] For recreational fishing, the direct losses could be taken to be the expenditures of the fishers plus the value of fish caught.[32-34]

In general, entire fisheries would not be contaminated by an accidental release. The extent and the location of significant pollution would depend strongly upon the hydrological characteristics of the waterbody(ies) under consideration. Nearly entire fishing areas might be either severely contaminated or else only slightly contaminated depending upon the local water currents and sediment behavior in the area.

In a moderate-sized lake or reservoir, it might be reasonable to simply destroy all aquatic life. (This is sometimes done by fish and game departments as a means of getting rid of undesirable species. Chemicals are available (e.g., rotenone) that carry this out at modest cost, but which are harmless to warmblooded animals.) It would then be necessary to wait — perhaps for several years — before restocking. If the entire aquatic population and the bottom sediments were removed afterwards, it would not be necessary to wait for sediment burial to cover the contaminated sediment and detritus, and restocking could be carried out as soon as the water itself became safe. (Sediment removal is addressed in Section F3.4.1.3.1.) Against the rather high cost of sediment removal would need to be balanced the long waiting time — up to several decades — for natural removal and burial processes. This method would be of little value if there were a continuing source of contamination. Furthermore, it would not be practicable for large waterbodies.

F3.4.1.2.3 Crops

In general, the restriction of water for irrigation purposes would not be important at most sites. Therefore, the total costs for irrigation pathway losses would usually be relatively small.

Costs for losses of crops due to contamination of irrigation sources would vary depending upon the type of crop under consideration and upon the percentage of water obtained from irrigation.[24,26] The size of the area withdrawn from irrigation would be very site dependent and could range from a negligible area to a substantial one. In general, alternative sources of irrigation water would not be available. The losses could range from being the entire crop in regions in which irrigation water supplied the majority of the water for growth to being just that portion of the crop due to the added yield in regions in which irrigation water was used only as a supplementary source of water.

If irrigation were prevented altogether, the direct loss would be the value of the crops which were not grown or else the added value of the crops which were not obtained. However, if land were irrigated with contaminated water, then the direct loss could increase significantly. It would be the value of the crops which could not be used or grown, plus the value of the soil which was contaminated.

F3.4.1.2.4 Evacuation and Relocation

The number of people to be evacuated or relocated generally would be small. Thus, the total costs associated with such activities usually would be negligible compared to other interdictive costs.[35]

No evacuation and relocation costs would be anticipated for liquid pathways, except perhaps for a very few individuals who might happen to be in the way of essential work. In the event that a release coincided with major flooding, evacuation might be performed because of the flooding itself, without regard to the radioactive release. It is possible, however, that the return of evacuated people might have to be delayed beyond the time they would be permitted to return for flooding alone.

F3.4.1.3 Environmental Cleanup

The most expensive type of pathway interdiction involves cleanup of the entire environment. All animal and plant populations ultimately stand to benefit from such actions. The cleanup procedures which are considered here are dredging of contaminated sediment and decontamination of land. Such restorative procedures usually require either cleaning or storing of the contaminated materials.

F3.4.1.3.1 Dredging

Dredging costs can vary according to the size of the area to be dredged and amount of material to be removed.[1,36] The cost of dredging and disposal of sediments contaminated by a meltdown accident would be somewhat dependent on the depth of the water, the distance to disposal sites, the strength of the currents, etc. However, the major variable would be the amount of material to be removed.

Because of both the very high cost and the lack of suitable equipment,[36] it is unlikely that a large section of an estuary or any other waterbody would be dredged. It is more probable that, if dredging were done, it would be done only in localized regions of the estuary (or other waterbody) in which the contamination was particularly severe ("hot spots").

F3.4.1.3.2 Decontamination of Land

The only land which would probably be contaminated through liquid pathways would be that which was either irrigated or accidently flooded with polluted water. The cost of the "decontamination" can be taken to be roughly equal to the cost of condemning the land; thus the decontamination cost is approximately the value of the land.[26,35]

The most likely candidate for land decontamination would be shorelines of polluted waterbodies.

These might require removal of any contaminated debris deposited along the shoreline. In addition, they might require removal of any fine-grained sediments either deposited from the waterbody or else just in temporary contact with the contaminated water, as from flooding or tidal action. Such decontamination might be needed periodically if there were a continuing source of contamination. Shorelines consisting of coarse nonadsorptive sediments, such as clean sands, would probably not require decontamination beyond removal of contaminated debris.

Another possible candidate for land decontamination would be floodplains of polluted waterbodies. These might require removal of any contaminated sediment and other materials deposited by flooding. In any case, disposal of the contaminated material would probably constitute the largest part of both the effort and the cost.

F3.4.2 Indirect Costs

The costs of pathway interdiction could involve any or all of those costs just listed plus indirect costs not mentioned previously. For example, if surface waterbodies were closed to general use, then all industries associated with the use of those bodies would suffer. Industries such as those related to recreational and commercial fishing could experience severe losses. The losses in such directly dependent industries would, in turn, generate other losses throughout the economy. The detailed estimation of these costs is beyond the scope of this project. It can be said that, in general, the obvious direct costs for pathway interdiction would probably be just a small portion of the total costs. Furthermore, there would be costs associated with the disruption of people's lives for which monetary estimates could not legitimately be made.

Indirect economic costs might be reduced by intervention; for example, unemployed fishers or others affected by the interruption of certain industries could be helped either to relocate or to turn to other lines of work. Indirect benefits from such intervention would also be possible. For example, the money spent in a community for interdiction could stimulate other sectors of the local economy. However, the outlay for interdiction could be assumed to have been made at the expense of some other sector, which would suffer an indirect loss.

The indirect costs associated with pathway interdiction can be roughly approximated using multiplier analysis. In such analysis, the indirect losses (or gains) generated by a given direct loss (or gain) are taken to be a function of the number of times that money from any loss (or gain) in income would have been (or is) respent in the total economy. The multiplier represents the ratio of the total losses (or gains) to the direct losses (or gains).[37] The average national multiplier for the period 1975-1978 is 3.83.[38] Therefore, the indirect costs would be approximately 3 times the direct losses if intervention were ignored.

F4 Summary

Interdiction of radioactive releases may be possible either by preventing contact between the radioactive material and accessible water, or by interrupting the flow of the radioactive material somewhere along its pathway to the human population. The former type of mitigating actions, which are taken close to the source of radioactivity, are denoted as source interdictive procedures, whereas the latter type of actions, which are taken along the exposure pathways, are denoted as pathway interdictive procedures.

Source interdiction is possible under many circumstances for the liquid pathways. (It is currently always impossible for atmospheric releases.) For example, contact with groundwater could often be prevented for decades or centuries by sealing off the melt debris by a grout curtain, or for several years by lowering the watertable at the site. Under some circumstances, neither method may be feasible.

Interdiction after the radioactivity leaves the reactor site but before it reaches any waterbodies contacted by the human population may also be possible in some situations. The most feasible method is by withdrawal of the contaminated water from the ground.

Interdiction farther along the pathway to the human population is more universally possible. However, the scope of the measures required to produce a given level of mitigation are much larger for such pathway interdiction than for the interdiction closer to the source. Furthermore, the efforts for any pathway interdiction might need to be carried on for decades. Consequently, for both these reasons, the costs for pathway interdiction are generally very much higher than those for source interdiction.

Pathway interdictive procedures can be divided into two categories: interruptive and restorative. The former category does not limit contamination of the environment; it merely attempts to prevent the interaction of the human population with the radioactivity. It includes proscribing or limiting the use of contaminated water supplies, limiting the use of contaminated waterbodies, confiscating contaminated aquatic life, preventing the use of polluted water for irrigation, and disposing of contaminated foodstuffs. The latter, restorative, category attempts to remove all contamination from the environment; it limits the interaction of many or all plant and animal species with the radioactivity. It includes dredging of tainted sediments, purification of polluted water and decontamination of shorelines and land. In general, the costs for restorative procedures are much higher than those for interruptive procedures.

Monitoring is a necessary concomitant of all interdiction methods. The cost is highly variable, depending upon the effort expended and upon the size of the area to be monitored.

Table F6 summarizes the effectiveness, feasibility, and cost of some interdiction methods. Cost estimates should be regarded as illustrative only.

Table F6. Interdiction Methods

Method of Interdiction	Class	Feasibility	Effectiveness	Time and Cost Required[a]
Installing Grout Curtain	Source	Nearly always possible; time required may be significant	Very effective for decades to centuries, perhaps longer	15 mos - 3 yrs; $10 x 10^6$ and up.
Lowering Watertable	Source	Possible under some conditions	Moderately effective for a few years to decades - requires continuous pumping	Requires a few months to implement; $1 x 10^6$ and up.
Pumping Contaminated Water from Aquifer	Source	Sometimes feasible	Moderately effective - requires continuous pumping	Requires a few months to implement - pumping for several years at $3 x 10^5$ per well; total $3 x 10^6$ - $30 x 10^6$; additional costs for storage and treatment of contaminated water.
Injecting Uncontaminated Water into Aquifer	Source	Sometimes feasible	Moderately effective - requires continual pumping	Same as above.
Substituting Drinking Water	Pathway	Generally possible	Locally effective	$1-800/cap./month, possibly lower; may have to be continued for many years.
Dredging	Pathway	May not be possible due to lack of enough suitable equipment	Locally effective	Requires years; $36,000/ha.; may have to be continued for years
Interdicting Fish	Pathway	Possible in localized waterbodies	Effectiveness may be limited because of mobility of some affected species	Lakes: $74,000/km^2; rivers: $154,000/km^2; coasts: $100,000/km; may have to be continued for many years.
Interdicting Land	Pathway	Possible for floodplains may be difficult for shorelines if contamination is continuous	Effective for floodplains; may be of limited effectiveness for shorelines	Cost of farm land: $420/ac (1040/ha) average; cost of urban land: $32,500/capita; disposal costs for contaminated materials would probably be more realistic than condemnation costs for the affected land; shoreline interdiction may be required for many years.
Relocating Inhabitants	Pathway	Possible but very seldom needed	Effective	$40/person/day

[a] The costs in this table are not necessarily meant to be representative of the range of costs which might actually be encountered; instead, they are taken from examples discussed in Addenda F4 and F5 and represent the costs for those specific situations.

References

[1] U.S. Nuclear Regulatory Commission, *Liquid Pathway Generic Study*, NUREG-0440, Office of Nuclear Reactor Regulation, USNRC, Appendix E, February 1978.

[2] U.S. Nuclear Regulatory Commission, "Revised Draft Environment Statement, Part III, A Supplement to the Generic Environmental Statement - Part II - Comparing the Overall Risk and Consequences Between Floating Nuclear Plants and Land-based Nuclear Plants Relative to Accidental Releases of Radioactivity to the Environment," NUREG-0127, Office of Nuclear Reactor Regulation, USNRC, May 1978.

[3] U.S. Nuclear Regulatory Commission, *Reactor Safety Study, An Assessment of Accident Risks in U.S. Commercial Nuclear Power Plants*, WASH-1400, USNRC Appendix VI, Chapter 8, October 1975.

[4] U.S. Nuclear Regulatory Commission, *Liquid Pathway Generic Study*, NUREG-0440, Office of Nuclear Reactor Regulation, USNRC, Chapter 7, February 1978.

[5] D. K. Todd and L. Huisman, "Groundwater Flow in the Netherland Coastal Dunes," Proc. ASCE, Hydr. Div., *85*:HY7, 63-81 (1959).

[6] Ad Hoc Committee on Technology of Drilling for Energy Resources, *Drilling for Energy Resources*, National Academy of Science, Wash, DC, 1976.

[7] S. N. Davis and R. J. M. DeWiest, *Hydrogeology*, John Wiley and Sons, New York, NY, 1966.

[8] Frank W. Stubbs, Jr., Editor, *Handbook of Heavy Construction*, McGraw-Hill New York, NY, Chapter 7, 1959.

[9] R. D. Statler and C. W. Gulick, Jr., Private communication with S. J. Niemczyk and B. Appel, Sandia Laboratories, Albuquerque, NM, January 1979.

[10] Robert Bowen, *Grouting in Engineering Practice*, John Wiley and Sons, New York, NY, 1975.

[11] BWR generalized diagram taken from *Grand Gulf Nuclear Station, Final Safety Analysis Report*; Mississippi Power and Light Company, Volume I.

[12] PWR diagram taken from Watts Bar, Final Safety Analysis Report, Mississippi Power and Light Company, Volume 1.

[13] L. W. LeRoy, D. O. Leroy and J. W. Raese, editors, *Subsurface Geology*, Colorado School of Mines, Golden, CO, 1977.

[14] U. S. Nuclear Regulatory Commission, "Draft Regulatory Guide," Safety-Related (Category 1), Permanent Dewatering Systems, USNRC, Attachment A, 1979.

[15] U.S. Nuclear Regulatory Commission, Reactor Safety Study, *"An Assessment of Accident Risks in U.S. Commercial Nuclear Power Plants,"* WASH-1400 (NUREG 75/014), USNRC, Appendix VI, October 1975.

[16] U.S. Nuclear Regulatory Commission, Reactor Safety Study, *"An Assessment of Accident Risks in U.S. Commercial Nuclear Power Plants,"* WASH-1400 (NUREG 75/014), USNRC, Appendix V, October 1975.

[17] H. W. Lewis, et al, *Risk Assessment Review Group Report to U.S. Nuclear Regulatory Commission*, NUREG/CR-0400, USNRC, 1978.

[18] U.S. National Committee on Radiological Protection, "Basic Radiation Protection Criteria," NCRP Report 39, NCRP Washington, DC, 1971.

[19] International Commission on Radiological Protection, "Recommendations of the International Commission on Radiological Protection (Adopted September 17, 1965)," ICRP Publication 9, Pergamon Press, Oxford, 1966.

[20] International Commission on Radiological Protection, "The Principles and General Procedures for Handling Emergency and Accidental Exposures of Workers," ICRP Publication 28, Pergamon Press, Oxford, 1978.

[21] U.S. National Committee on Radiation Protection, "Permissible Dose from External Sources of Ionizing Radiation," NBS Handbook 59, National Bureau of Standards, Washington, DC, 1962.

[22] U.S. National Committee on Radiological Protection, "A Review of the Current State of Radiation Protection Philosophy," NCRP Report 43, NCRP, Washington, DC, 1975.

[23] R. D. Statler, Private Communication with B. Appel, Sandia Laboratories, Albuquerque, NM, October 1978.

[24] W. Stone, private communication with B. Appel, Ventura County Flood Control and Water Resources, Ventura, CA, October 1978.

[25] D. K. Todd, Editor, *Water Encyclopedia*, Water Information Center, Inc., Huntington, NY, 1959.

[26] W. Lerner, Editor, Statistical Abstract of the United States, *1977*, U.S. Department of Commerce, Washington, DC, 1977.

[27] A. D. Parker, *Planning and Estimating Underground Construction*, McGraw-Hill, New York, NY, 1970.

[28] U.S. Nuclear Regulatory Commission, Nuclear Regulatory Guide 1.109, "Calculation of Annual Doses to Man from Routine Releases of Reactor Effluents for the Purpose of Evaluating Compliance with 10 CFR Part 50, Appendix I," USNRC, Washington, DC, March 1976.

[29] Manager of Southwestern Public Service Co., Private Communication with B. Appel, Albuquerque, NM, November 1978.

[30] Arapaho Drilling Co., Private Communication with B. Appel, Farmington, New Mexico, November 1978.

[31] T. Tayes, Private Communication with B. Appel, Farmington, NM, November 1978 and February 1979.

[32] *Fisheries of the United States*, 1975, Current Fisheries Statistics No. 6900, NOAA, 1977.

[33] Anonymous, *Economic Activity Associated with Marine Recreational Fishing*, U.S. Fish and Wildlife Service, 1972.

[34] National Analysts, "1975 National Survey of Hunting, Fishing, and Wildlife Associated Recreation," U.S. Fish and Wildlife Service, 1972.

[35] U.S. Nuclear Regulatory Commission, *Reactor Safety Study, An Assessment of Accident Risks in U.S. Commercial Nuclear Power Plants*, WASH-1400, USNRC, Appendix VI, Chapter 12, October 1975.

[36] U.S. Army Corps. of Engineers, "The EPA Kepone Mitigation Feasibility Project Report," U.S. Environmental Protection Agency, Appendix B, 1978.

[37] H. S. Sloan and A. J. Zurcher, *Dictionary of Economics*, Barnes & Noble, New York, NY, 1970.

[38] Council of Economic Advisers, "Economic Indicators," Govt. Printing Office, April 1979.

ADDENDUM F1

Calculation of Drawdown

The feasibility of lowering the watertable by pumping depends on several factors, including the soil conditions, especially the permeability, the distance to any significant surface waterbody, and the depth to a continuous impervious substratum. In investigating the usefulness of lowering the watertable in the vicinity of the melt, it is useful to consider the amount of drawdown which occurs in any substratum for given amount of pumping. In this addendum, the amount of drawdown resulting from pumping at wells evenly spaced on a circle centered on the reactor is considered.

If a confined aquifer underlies the reactor, as shown in Figure F3a (presented in Subsection F2.1.2 of this appendix), the drawdown at each well due to its own discharge is

$$h_o - h_w = \frac{Q}{2\pi Kb} \ln \frac{r_o}{r_w} , \qquad \text{(F1-1)}$$

where Q is the pumping rate in gallons per day (gpd), K is the permeability of the aquifer in gpd/ft^2, and other symbols are as indicated in Figure F3a. If there are n wells, each pumping Q gpd, then the total discharge is nQ, which must be made up by recharge from the aquifer. If recharge takes place on a front whose width is n times the circumference of each circle of radius r_o, then the total recharge is

$$nQ = 2\pi n r_o bKd , \qquad \text{(F1-2)}$$

where d is the hydraulic gradient. From Equation (F1-2), we have

$$r_o = \frac{Q}{2\pi bKd} . \qquad \text{(F1-3)}$$

This value of r_o can now be used in Equation (F1-1) to determine the drawdown at each well. If the wells are equally spaced on a circle of radius r_c centered on the reactor, the drawdown at the reactor centerline is

$$h_o - h_c = n(h_o - h_w) \frac{\ln \frac{r_o}{r_c}}{\ln \frac{r_o}{r_w}} . \qquad \text{(F1-4)}$$

The drawdown at the ith well, due to its own and all other wells' pumping, is

$$(h_o - h_{wi}) = (h_o - h_w)\left[1 + \sum_{\substack{j=1 \\ j \neq i}}^{n} \frac{\ln\left(\frac{r_o}{d_{ij}}\right)}{\ln\left(\frac{r_o}{r_w}\right)} \right] , \qquad \text{(F1-5)}$$

where d_{ij} is the distance from the ith well to the jth well.

For the more usual case of an unconfined aquifer under the reactor (Figure F3b), the drawdown at each well due to its own pumping is

$$h_o - h_w = h_o - \left[h_o^2 - \frac{Q \ln\left(\frac{r_o}{r_w}\right)}{\pi K} \right]^{1/2} . \qquad \text{(F1-6)}$$

In this case, r_o is given by

$$r_o = \frac{Q}{2\pi h_o Kd} . \qquad \text{(F1-7)}$$

The drawdown under the reactor is again given by Equation (F1-4) and the total drawdown at each well by Equation (F1-5).

Special attention must be given to leaky aquifers, or to cases when the well does not completely penetrate the aquifer, as shown in Figure F3c.

After the total drawdown at each well is computed for various numbers of wells sized to give the desired drawdown at the location of the melt debris, one can choose a combination based on cost and whether the aquifer can adequately recharge the total flow. Some combinations of permeability, aquifer thickness, and hydraulic gradient may totally preclude the possibility of achieving the necessary drawdown. In other cases, the aquifer may be incapable of the necessary recharge. All illustrative example is given in Subsection F2.1.2.

ADDENDUM F2

Calculation of Near-Field Doses and Dose Rates

F2.1 Details of Calculations

The computer code which implements the Reactor Safety Study Consequence Model[15] has been used to compute the near-field dose resulting from radionuclide deposition from the atmospheric release. The RSS Consequence Model considers seven categories of atmospheric release for melt-through in PWR's and four for melt-through in BWR's, with the various categories differing in severity with respect to atmospheric consequences. Only PWR releases have been considered here for computing near-field contamination. The probabilities of releases occurring in each category have been previously computed.[16] However, there is a considerable uncertainty associated with the probabilities.[17] The median probabilities from Reference 16 have been used, recognizing that significant uncertainties exist.

The RSS Consequence Model Program (CRAC) combines the release combinations with weather conditions and dosimetric data to determine doses to various organs from the radioactive material in the passing cloud and from material deposited on the ground. It can be assumed that no mitigating action would be undertaken during cloud passage, so that only the ground dose is considered here.

CRAC divides the area surrounding the reactor into a radial grid. The model was principally intended to calculate effects at offsite locations. Therefore, the midpoint of the first spatial interval is at 402 meters (1/4 mile) from the release point. If the distance between the reactor and the closest ground-deposited radioactivity is less than the 402 meters, contamination is shown for the first spatial interval.

Atmospheric conditions are site dependent. In order to keep the computations within reasonable bounds, a single generic site — typical of a northeastern river valley — was chosen for analysis. Because rainfall probability is the most important meteorological variable here, the results of these computations are conservative for arid sites and unconservative for very wet sites.

Computations have been carried out for all combinations of the following conditions:

1. three PWR release categories, including the two sub-categories of the most severe category, a mid-range category, and a relatively benign release category;
2. two conditions of atmospheric stability, one stable and one unstable;
3. two wind velocities, one low (0.5 m/s) and one high (5 m/s); and
4. two precipitation conditions, one with rainfall near the reactor and one without rainfall.

No shielding was included and resuspension of deposited radioactivity was ignored.

Ground doses to the bone marrow, lung, regenerative cells of the lower large intestine, and whole body for the first spatial interval were inspected. The bone marrow dose rate was somewhat larger in most cases, and this has been taken as a proxy for all other doses*. The results of the calculations are presented in the next section of this addendum.

F2.2 Results of Calculations

The dose rates can be displayed in terms of complementary cumulative distribution functions (CCDF's) of the rates. Such CCDF's display the possibility of equaling or exceeding a specified dose rate. The CCDF's for the bone marrow dose rate** due to ground contamination at the time of cloud passage and at 7 days, 30 days, 100 days, and 1 year after passage are given in Figure F7 for the vicinity of the reactor. The dashed curve in that figure shows the

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*There is generally less than 20% difference between calculated bone marrow dose and whole body dose. Therefore, all conclusions relating to bone marrow can be applied to whole-body dose with small error.

**See footnote for the previous subsection.

CCDF for the dose rate at one year with a decontamination factor of 100, that is, with ground cleanup sufficient to reduce dose rate to 0.01 times its value before cleanup. (This magnitude of decontamination would probably not be practical for large land areas. However, in view of the rather small area contaminated in the immediate vicinity of the reactor, a factor of 100 is believed possible.) This curve is included to indicate the minimum dose rate to be expected at the time source interdiction operations begin in the vicinity of the reactor.

[redacted]

ADDENDUM F3

Dose and Dose Rate Limits

F3.1 Occupational Dose Limits for Emergencies

There are no absolute limits for occupational exposure in accident situations. The criteria which would be employed after a core-melt accident could be expected to be extremely dependent on the actual accident conditions, in particular, on the potential for significant early exposure of the population-at-large.

The recommended emergency occupational dose limits are generally based on the philosophy that such an exposure for an individual would be a once in a lifetime occurrence. Limiting dose recommendations of the NCRP for individuals are presented in Table F3.1.[18] The NCRP limit for non-lifesaving emergencies is 25 rems of whole-body dose, while that for lifesaving emergencies is 100 rems. Other recommended values for emergency situations are comparable.[19-22]

Table F3.1 NCRP Limiting Dose Recommendations[18]

Emergency - Lifesaving	
Individual (older than 45 yrs.)	100 rem
Hands and forearms	200 rem additional
	300 rem total
Emergency - Non-Lifesaving	
Individual	25 rem
Hands and forearms	100 rem
	125 rem total

All of these doses are much in excess of the dose permitted by Part 20 of Title 10 of the Code of Federal Regulations for nonemergency occupational exposures. 10 CFR 20 requires that an individual receive no more than 3 rem of whole-body dose in less than 3 months as a result of occupational exposure; and this dose is permitted only if an individual's accumulated dose over time is not too large. Otherwise, a maximum individual dose of only 1-1/4 rem of whole-body dose is permitted in 3 months. Potential exposure of an individual to a larger dose than that permitted by 10 CFR 20 might require an NRC waiver.

F3.2 Nonoccupational Dose Limits for Controlled Releases

The commonly-used intervention criteria for nonoccupational exposures[18] resulting from controlled releases are given by 10 CFR 20 and are based on the concentrations of the radionuclides in the relevant portions of the exposure pathways. Specifically, intervention is thought to be warranted if the concentrations are such that more than 0.50 rem of whole-body dose would be received by any individual in the population or more than 0.17 rem of whole-body dose would be received by an average individual in the population; both limits are for any one year period and are for radiation from all nonmedical sources.[18]

Inasmuch as two or three liquid pathways might be significant at a given site, the average individual dose from each pathway should probably be no more than one-half or one-third of 0.17 rem per year, respectively, unless the exposed populations are different. If atmospheric pathways are important, then the acceptable average dose per liquid pathway should be reduced even further. The acceptable concentrations for each pathway can be determined from the maximum permissible concentrations (MPC's) for nonoccupational exposure by taking into consideration that more than one radionuclide and more than one pathway can both contribute to the dose.

F3.3 Nonoccupational Dose Limits for Accidental Releases

In contrast to the controlled release situation, the commonly-used intervention criteria for nonoccupational exposures resulting from accidental releases are given by FDA guidelines for ingestion and by EPA Protective Action Guides (PAG's) for all other modes of exposure. Both are based on the projected whole-body dose which will be received by an individual as a result of the accident. For example, if the projected whole-body dose is above 1 rem, then according to the EPA PAG's mitigating action should be considered; and if the projected dose is greater than 5 rem, then some protective action must be taken. The FDA criteria for ingestion are generally lower and more complicated than the EPA ones.

ADDENDUM F4

Detailed Source Interdiction Costs

F4.1 Introduction

The costs presented in this addendum are intended to be illustrative and are not necessarily representative. Furthermore, it is not to be expected that the indicated ranges of costs are all-inclusive.

F4.2 Wells

Estimates for installation of wells vary from approximately $40,000 per well[23,24] up to several hundred thousand dollars. A breakdown of costs per well was obtained for one Sandia project in which 21 holes were drilled in sandstone, anhydrite, gypsum, and salt.[23] Each hole was approximately 6 inches in diameter and was drilled to a depth of 1800 feet. Costs included:

a. Site preparation - includes 10 miles of roads, drill pads 50 ft x 50 ft x 21 ft	$115,000[*]
b. Total rig costs - four rigs, plus operators	350,000
c. Material costs - casing, mud, chemicals, bits	75,000
d. Support services - cementing, logging, cores, water	120,000
e. Engineering management	130,000
TOTAL	$790,000

The drilling costs for this project amount to approximately $5480 per hole, plus $17.90 per foot, at 1975 costs. These costs are estimated to be $6000 per hole plus $19.60 per foot at 1977 prices.[**,†] The additional work and trouble required for rather precise slant-hole drilling is estimated to increase the footage costs by 20 percent, resulting in $23.50 per foot.

Costs for dewatering wells are carried out using $19.60 per foot. For a 250-foot dewatering well, the initial cost is thus $10,900. This figure does not include the cost of handling the pumped water; however, this should be small in comparison with other costs if the water is not contaminated. Under adverse circumstances, it is estimated that the per-foot costs could increase by 20 percent, so that each well would cost $13,100 to drill. (By comparison, drilling costs from Reference 25, brought forward to 1977 costs, are approximately $9,200/well.) These costs for wells do not include the cost of treatment of any contaminated water.

For a dewatering well 250 feet deep, the initial costs are estimated to be as follows:

Drilling Costs	$10,900
Other Capital Cost	30,100
Total Initial Cost	$41,000

The annual costs are estimated to be:

Operating Costs	$1,000/yr
Service Costs	800/yr
Energy Costs	43,100/yr
Total Annual Costs	$44,900/yr

If pumping is continued for 10 years, the present value of annual cost at 10 percent interest is $276,000. The total cost per well, referred to present time is:

Present Value of Annual Cost	$276,000
Initial Cost	41,000
Total Equivalent Cost	$317,000

It can be seen that the initial cost is only a small fraction of the total cost. In this example, three dewatering wells would have a present cost of approximately $\$1 \times 10^6$. Note that this is merely an illustrative example, and does not show the very wide range of costs that might apply.

The pumped water might require treatment. Reference 25 gives investment costs and total costs for treatment of both surface water and groundwater in 1964. The treatment of surface water in that reference includes softening, chemical coagulation, sedimentation, rapid sand filtering and chlorination. This level of treatment, plus extensive ion exchange, is expected to be appropriate for treatment of slightly contaminated water. The treatment of groundwater in that reference includes iron removal, softening, and chlorination. This level of treatment is approximately appropriate for uncontaminated water to be used for aquifer recharge.

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[*] Figures represent 1975 prices.

[**] Costs are generally referred to 1977 levels in this report. The reason is that the indexes for carrying costs to 1978-79 were not available, in some cases, at the time of writing.

[†] These costs are in line with those for the national averages drilling for oil and natural gas. In 1973, these costs averaged $22.50 per foot including all costs,[6] whereas the Sandia project averaged $20.90 per foot in 1975.

Total costs average approximately 50 percent for amortization of investment costs and 50 percent for operating costs (20% for personnel, 17% for energy, 7% for chemicals, and 6% for miscellaneous). The 1977 costs* for water treatment can be fitted by the equations:

$$I_S = 682\ Q^{.651}\ ,$$
$$I_G = 310\ Q^{.633}\ ,$$
$$C_S = .230\ Q^{-.354}\ ,$$
$$C_G = .096\ Q^{-.374}\ ,$$

where I_S is investment cost for treatment of surface water, I_G is investment cost for treatment of groundwater, both in thousands of dollars, C_S is total cost for treatment of surface water, C_G is total cost for treatment of groundwater, both in dollars per 1000 gallons, and Q is throughput in 10^6 gpd (gallons per day).

For the numerical example considered in Subsection F2.1.2, in which pumping of approximately 10^6 gpd of uncontaminated water was required to lower the watertable, the investment cost for the required facilities would be approximately $310,000 and the corresponding operating cost would be approximately $35,000 per year. If the water became somewhat contaminated by radioactive materials, the investment cost would rise to $682,000 and the total operating cost would be $84,000 per year.

The initial investment costs plus the present values of the annual costs not attributable to investment (for 10 years treatment time) are:

	Uncontaminated	Contaminated
Initial Cost	$310,000	$682,000
Present Value of Annual Cost	108,000	258,000
Total Equivalent Cost	$418,000	$940,000

In this example, the three wells would have a present equivalent cost equal to approximately $1 x 10^6, so that treatment costs could range from an amount less than half of pumping costs to an amount approximately equal to pumping costs. Additional costs would be incurred due to disposal of contaminated water treatment media.

The cost for treatment of uncontaminated water would be reduced slightly if only chlorination were required. On the other hand, the cost for treatment of contaminated water could be much higher than the above estimates if the water were extensively contaminated. An example of severe contamination would be a sudden release of sumpwater into an aquifer.

F4.3 Grouting

The same drilling example can be used to consider grouting costs for 520 holes, each hole 110m (361 ft) long (Subsection F2.1.3). Using the cost figures which account for the required slant drilling, the cost per hole is $14,500. To this must be added the cost of grouting:[27]

Grout	$4.00/ft^3
Labor	$2.30/ft^3
Cement	$1.25/sack (approx. 62.5¢/ft^3).

Each hole is assumed to require 780 ft^3 of grout, so the grouting cost is $5,400 per hole. The total cost per hole is

Drilling	$14,500
Grouting	5,400
Total Cost	$19,900

The total cost for all 520 holes is thus approximately $10 x 10^6. If a three-row approach were required instead of a single-row one, all these grouting costs could be doubled. To these costs must be added the dewatering costs, the costs of sealing off the top of the debris, and the costs of radiological controls.

It is again emphasized that these cost estimates are illustrative only. It is possible that the total cost might run somewhat below this figure; it is also possible that it might exceed it by a large factor.

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*These costs in Reference 25 have been referred to 1977 costs, using the EPA Sewage Treatment Plant Cost Index for investment costs and the Implicit GNP Deflator for operating costs.[26]

ADDENDUM F5

Detailed Pathway Interdiction Costs

F5.1 Introduction

The pathway costs presented in this addendum are intended to be only illustrative. Examples are presented for monitoring, as well as for various pathway interruption and restoration prodecures.

F5.2 Monitoring

As an example of monitoring costs, assume that a contaminated aquifer is expected to discharge into a lake. The site is close to a small community, where housing is available. No laboratory facilities are locally available, and a small laboratory must be constructed at the site. This laboratory would handle routine analysis; more complex jobs would be sent out for laboratory work.

A total of 20 wells might be drilled into the aquifer; 15 of these would withdraw water for testing, and 5 would return the water to the aquifer. The test wells would be operated part time; the recharge wells would operate full-time. Temporary tankage and minimal treatment (chlorination) would be provided.

Initially, 50 employees might be used for monitoring and analysis, at a cost (including overhead and support) of $90,000/person-year. Boats for water, fish, and sediment sampling might be rented at a cost of $90,000 per year. Portable monitoring equipment would be provided. The cost of laboratory operation, replacement of damaged and worn-out equipment, and contract laboratory work would be $300,000 per year.

The level of the monitoring effect could presumably be reduced after some time if appropriate source interdictive measures were taken. Consequently, monitoring in this example is assumed to continue at the initial scale for five years, to then be cut back by 50 percent for 3 years, to again be cut by 50 percent for 5 more years, and to continue at a minimum level of $180,000 per year for an additional 5 years.

The initial costs for this example are as follows:

Well drilling	$ 820,000
Laboratory construction	1,500,000
Water storage and treatment	1,000,000
Portable Equipment	250,000
Total Initial Cost	$3,570,000

The related annual costs at full operation are:

Pumping	$ 562,000
Treatment	50,000
Boat leasing	90,000
Personnel	450,000
Lab operation	300,000
Total	$1,450,000

The present value of annual costs, at 10 percent interest, are approximately

5 years at $\$1.452 \times 10^6$	$5,500,000
3 years at $\$.726 \times 10^6$	1,120,000
4 years at $\$.363 \times 10^6$	786,000
3 years at $\$.180 \times 10^6$	143,000
Total Present Value	$7,550,000

The total of initial cost plus present value of annual cost for the example is estimated to be slightly over $\$11 \times 10^6$.

This example is intended to be illustrative only, and does not take into account the possible variation in local conditions. Consequently, the actual costs could be either much higher or much lower than those indicated.

If the environmental program just described were instituted, this would probably include some monitoring of fish. However, if a valuable sport or commercial fishery existed in the area or in downstream reaches, enhanced fish monitoring might be deemed necessary. Because of the mobility of some finfish, this could extend far beyond the area covered by water quality monitoring. Annual costs, for a maximum fish-monitoring effort at a moderate-sized lake, are estimated to be:

Personnel	\$450,000
Laboratory work	300,000
Boats and equipment	200,000
Total	\$950,000

As with water-quality monitoring, costs could probably be reduced after a few years if adequate source interdictive procedures were taken. It is assumed in this example that a well-equipped laboratory already exists, so that no capital costs are required. If laboratories for fish analysis must be built and equipped, an initial capital cost of \$1.5 x 10^6 for each laboratory is estimated.

The cost of monitoring would depend to a great extent on the size of the fish population exposed and the estimated danger to humans. If a large fish population were exposed, and the fish were extensively caught and consumed, the monitoring effort and related cost would be higher.

F5.3 Pathway Interruption

F5.3.1 Drinking Water

Cost estimates for shipping water were obtained from various sources. A private New Mexico firm charges 50c/gallon for 1 gallon jugs of water, plus \$1/mile for round trip expenses. The truck in this case can hole 250-300 cases, with 6 gallons/case.[29] Another firm charges \$28/hour for a truck which holds 100 barrels (3200 gallons).[30] The National Guard has 10-ton tractors to haul either 5,000 or 4,400 gallon trailers. The only costs here are pay allowances for two men at a rate of approximately \$85 per day and fuel for the truck.[31]

As an example, the following hypothetical situations might be considered. A small community of approximately 150 has the aquifer which is the source of all of the town's water severely contaminated so that all water has to be supplied from other sources. The total water usage per person would most likely be cut back, say to 50 gallons/day (175 ℓ/day) from the average 200 gallons/day (700 ℓ/day). Therefore, in one month the entire population of this community would need 225,000 gallons of water.

The usual emergency procedure for the National Guard would be to fill a pumper truck, haul it to the community, and fill its storage tank. If two 5,000 gallon capacity trucks were available, they would need to make 24 trips each to supply the water used in a month. The cost would be \$85 per truck (for the 2 men) x 2 trucks x 24 trips = \$4,080 to haul the water, plus the cost of fuel. That cost would be \$.80 per gallon x 12.5 gallons per 50 mile round trip x 2 trucks x 24 trips = \$480 for fuel. Therefore, the National Guard's total cost would be approximately \$4,600 for one month.

For a private firm that could supply gallon jugs of water at \$.50/gallon, the cost of supplying water for one month would be much higher: \$112,500 for the water alone. The truck, which can only hold 1800 gallons, would have to make 125 trips. At \$1/mile expenses, the total transportation cost would be \$6250. Therefore, the private firm's total cost to supply water to the same community for one month would be approximately \$119,000. Thus, the estimated total costs for supplying this water range from \$5000 to \$120,000, depending upon the supplier;[29-31] these costs range from \$35 to \$800 per capita per month.

If this same hypothetical situation existed near a large lake and only drinking water needed to be supplied, then for a month only 1215 to 2430 gallons (4500 to 9000ℓ)water would need to be shipped in for the community. The National Guard's costs would be approximately \$95 for one month. The private firm's costs would be about \$708 to \$1315 to supply the same water. Thus, the estimated costs for supplying this amount of water vary from \$100 to \$1500, again depending on the supplier; these costs range from less than \$1 to about \$10 per capita per month.

These estimates for a community of 150 should be considered to provide a maximum cost per person. For a larger community, more economical transport means (railroad tankers, pipelines, tank ships, etc.) could be utilized.

At sites at which significant drinking water contamination occurred, many more than just 150 people would generally be affected. For some surface waterbodies, millions of people could even be impacted. Thus, even though the costs per person might be much less than the estimates presented here, the total costs for substitution of water for a given site might be orders of magnitude more than those indicated for the examples.

As an alternative to substitution, water treatment might be considered. Such costs are dealt with in Addendum F4. For example, minimal treatment (iron removal, softening, and chlorination) of 10^6 gallons per day (enough for 5000 people) would require a capital investment of approximately \$300,000 and an operating budget of \$35,000 per year. More extensive treatment (minimal treatment plus chemical coagulation, sedimentation and rapid sand filtration) would approximately double both these costs. If there were existing facilities, the only investment costs necessary might be those for improvement of existing facilities. In any case, additional costs would be incurred due to handling and disposal of radioactive processing media.

F5.3.2 Finfish and Shellfish

The total values of the commercial and recreational finfish catches in 1975 for various important U.S. fisheries are given in Table F5.1.[32-34] It is often useful to consider the value of a fishery as a function of either shoreline length or surface area of the relevant waterbody. The values of some important U. S. fisheries as functions of these variables are also given in Table F5.1.

Most fish tend to be taken from densely populated fishing grounds; this is particularly true for shellfish. Therefore, distribution of the catch uniformly over a fishery can be somewhat misleading. It has been done only to provide an estimate of the possible economic consequences.

Table F5.1 Values of 1975 Finfish and Shellfish Catches for Some Important U. S. Fisheries in Billions of Dollars per Year[a]

Region	Value of Commercial Landings[b] ($\$10^9$)	Value of Recreational Landings[c] ($\$10^9$)	Amount Spent by Recreational Fishers ($\$10^9$)	Total Value ($\$10^9$)	Length of Tidal Shoreline[d] (km)	Length of Coastal Shoreline[d] (km)	Average Distributed Value[e] ($\$10^9$)
North Atlantic	0.150	0.104	0.682	0.936	9,860	761	95,000/(km-ts); 1,230,000/(km-cs)
Middle Atlantic	0.104	0.059	0.736	0.899	17,553	689	51,000/(km-ts); 1,300,000/(km-cs)
South Atlantic	0.061	0.112	0.646	0.819	19,187	1,880	43,000/(km-ts); 936,000/(km-cs)
Gulf of Mexico	0.271	0.093	1.440	1.804	27,580	2,625	65,000/(km-ts); 687,000/(km-cs)
California Coast	0.129	0.017	0.241	0.387	3,427	523	113,000/(km-ts) 740,000/(km-cs)
Oregon and Washington Coast	0.085	0.029	0.367	0.481	4,436	728	108,000/(km-ts); 661,000/(km-cs)
Great Lakes[f]	0.125	0.022(g)	0.424(g)	0.571	-	-	(h)
Lake Superior	0.009	0.004(g	0.074(g)	0.087	-	-	(h)
Lake Michigan	0.060	0.006(g)	0.116(g)	0.182	-	-	(h)
Lake Huron	0.005	0.002(g)	0.040(g)	0.047	-	-	(h)
Lake Erie	0.048	0.009(g)	0.172(g)	0.229	-	-	(h)
Lake Ontario	0.003	0.001(g)	0.022(g)	0.026	-	-	(h)
Streams and Rivers	0.016(i)	0.019	2.038	3.073	(j)	-	154,000/km^2
Reservoirs	-	0.171	3.143	3.314	(j)	-	55,000/km^2
Other Lakes, Etc.	-	0.171	4.246	4.417	(j)	-	74,000/km^2
Total	0.941	0.797	14.963	16.701	-	-	-

[a] The magnitudes of the catches are taken from the tables in Appendix E.

[b] The value of the commercial catch has been taken from *Fishery Statistics of the United States*.

[c] The value of the recreational catch has been estimated using the ex-vessel values of the combined Mississippi River and Great Lakes (except Lake Michigan) commercial catches, that is, $0.60/kg.

[d] Shoreline lengths are taken from Table E21 in Appendix E; the definitions of tidal shoreline and coastal shoreline are as given there.

[e] ts denotes tidal shoreline; cs denotes coastal shoreline.

[f] Great Lakes values indicate both Canadian and U. S. catches.

[g] The values of the recreational catch and the amount spent by recreational fishers have been distributed over the individual Great Lakes in proportions to the weight of the recreational catch.

[h] The values of the Great Lakes fisheries have not been distributed on an areal basis because it is believed likely that at least an entire lake would be adversely affected; there would be little point in interdicting a portion of a lake because of both the high mobility of fish and the rapid dispersal of the contaminant throughout the lake.

[i] Mostly from the Mississippi River and tributaries; includes some reservoirs.

[j] Not applicable; in the contiguous 48 states the area of all reservoirs and man-made ponds is approximately 6×10^4 km^2, the area of all major streams and rivers is 2×10^4 km^2 and the area for all lakes (other than the Great Lakes) and natural ponds is 6×10^4 km^2.[25]

F5.3.3 Crops

The nationwide (1977) averages for the value of crops from different types of farms are[26]

Grain	$127/acre
Cotton	157/acre
Tobacco	203/acre
Vegetables	691/acre
Fruit/nut	567/acre
Livestock	76/acre
Dairy	178/acre
All Farms	$135/acre

These figures are average values, and should be used with caution. For example, figures for Ventura County, California[24] are

Citrus	$1125/acre
Avocadoes	1400/acre
Vegetables	1740/acre

Note that these are all more than double the national averages. By the same token, exceptionally marginal land could produce considerably less. However, such marginal land is typically not irrigated.

F5.3.4 Evacuation and Relocation

Per capita costs for temporary relocation are estimated to be (per additional day)[35]

Lodging	$10
Food	6
Loss of income	19
Inconvenience payments	5
Total	$40/person-day

This estimate could vary up or down by 25 percent due to regional differences.

F5.4 Environmental Restoration

F5.4.1 Dredging

Estimates of dredging costs were obtained from a preliminary analysis of dredging kepone-contaminated sediment from the James River.[36,*] These costs are

Dredging	$5.63/$m^3$
Disposal	1.63/m^3
Total	$7.26/$m^3$

The estimated costs do not include the additional costs which might be incurred because of handling radioactive material. Such considerations could result in a total cost of approximately $7-10 per m^3 of material dredged.[1]

As an example, if an estuary 5 km wide and 50 km long were to be dredged to a depth of 0.1 m, the total amount of material to be removed would be 2.5×10^7 m^3, and the total cost would be \$180 $\times 10^6$. As for the proposed dredging of the James River Estuary, it is doubtful that appropriate equipment would be available to complete this task on a reasonable time scale.

F5.4.2 Decontamination of Land

The average value per acre of farmland is approximately $420.[26] It would cost approximately $230 per acre to decontaminate such land.[35] The value of developed property, per capita, is estimated to be $35,200.[26] For decontamination by a factor of 2, it would cost approximately $680 per acre for decontamination of that land.[35]

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[*] That analysis was not strictly applicable to the dredging requirements in the event of radioactive contamination, in that the depths and the areas dredged are not necessarily representative. Furthermore, the project analyzed was of such an enormous scope that sufficient equipment could not have been made available to complete the task on a reasonable time scale. However, it is believed that the unit costs computed for the James River Project are still reasonably applicable to nuclear decontamination.

APPENDIX G

Detailed Results of Calculations

Table G1.

Resultant Population Doses for the Nearshore Region of a Large Lake for Releases from the Melt Debris After a PWR-7 Accident[a]

t_{H_2O} = 1 day

Relative Leach Rate[b]	Pathway	Organ					
		Whole Body	Lung	Total Marrow	Skeleton	LLI	Thyroid
10^5	sh	1.3 + 5[c]	1.3 + 5	1.3 + 5	1.3 + 5	1.3 + 5	1.3 + 5
	dw	4.0 + 5	1.9 + 4	1.6 + 6	3.6 + 6	4.5 + 6	7.3 + 5
	af	8.2 + 4	3.2 + 4	2.6 + 5	3.4 + 5	2.4 + 5	1.1 + 5
	total	6.1 + 5	1.8 + 5	1.9 + 6	4.1 + 6	5.8 + 6	9.7 + 5
10^3	sh	1.1 + 5	1.1 + 5	1.1 + 5	1.1 + 5	1.1 + 5	1.1 + 5
	dw	3.9 + 5	1.7 + 4	1.5 + 6	3.6 + 6	4.9 + 6	7.1 + 5
	af	8.1 + 4	3.2 + 4	2.6 + 5	3.4 + 5	1.2 + 5	1.1 + 5
	total	5.9 + 5	1.6 + 5	1.9 + 6	4.0 + 6	5.1 + 6	9.3 + 5
10^1	sh	2.2 + 4	2.2 + 4	2.2 + 4	2.2 + 4	2.2 + 4	2.2 + 4
	dw	2.4 + 5	8.9 + 3	1.0 + 6	2.7 + 6	6.1 + 5	2.0 + 5
	af	7.0 + 4	2.9 + 4	2.2 + 5	2.7 + 5	5.9 + 4	9.4 + 4
	total	3.3 + 5	6.0 + 4	1.3 + 6	3.0 + 6	6.8 + 5	1.4 + 5
10^0	sh	5.0 + 3	5.0 + 3	5.0 + 3	5.0 + 3	5.0 + 3	5.0 + 3
	dw	7.4 + 4	4.8 + 3	3.2 + 5	8.2 + 5	1.0 + 5	4.4 + 4
	af	4.0 + 4	1.6 + 4	1.2 + 5	1.1 + 5	2.7 + 4	6.7 + 4
	total	1.2 + 5	2.6 + 4	4.4 + 5	9.3 + 5	1.3 + 5	1.2 + 5
10^{-1}	sh	1.5 + 3	1.5 + 3	1.5 + 3	1.5 + 3	1.5 + 3	1.5 + 3
	dw	4.1 + 3	1.5 + 3	1.3 + 4	1.3 + 4	2.9 + 3	8.1 + 3
	af	1.2 + 4	5.2 + 3	3.6 + 4	1.8 + 4	6.8 + 3	2.4 + 4
	total	1.7 + 4	8.1 + 3	5.1 + 4	3.2 + 4	1.1 + 4	3.4 + 4

t_{H_2O} = 10 days

Relative Leach Rate	Pathway	Organ					
		Whole Body	Lung	Total Marrow	Skeleton	LLI	Thyroid
10^5	sh	1.3 + 5	1.3 + 5	1.3 + 5	1.3 + 5	1.3 + 5	1.3 + 5
	dw	2.8 + 5	1.3 + 4	1.1 + 6	2.2 + 6	5.3 + 6	5.8 + 5
	af	3.3 + 4	7.1 + 3	1.2 + 5	2.0 + 5	1.4 + 5	2.10 + 5
	total	4.4 + 5	1.5 + 5	1.3 + 6	2.5 + 6	5.6 + 6	7.4 + 5
10^3	sh	1.1 + 5	1.1 + 5	1.1 + 5	1.1 + 5	1.1 + 5	1.1 + 5
	dw	2.7 + 5	1.1 + 4	1.0 + 6	2.2 + 6	4.9 + 6	5.7 + 5
	af	3.3 + 4	7.1 + 3	1.2 + 5	2.0 + 5	1.3 + 5	2.7 + 4
	total	4.2 + 5	1.3 + 5	1.3 + 6	2.5 + 6	5.1 + 6	7.0 + 5
10^1	sh	1.2 + 4	1.2 + 4	1.2 + 4	1.2 + 4	1.2 + 4	1.2 + 4
	dw	1.6 + 5	2.6 + 3	7.0 + 5	1.8 + 6	7.0 + 5	1.6 + 5
	af	2.9 + 4	6.8 + 3	1.0 + 5	1.7 + 5	3.2 + 4	2.4 + 4
	total	2.0 + 5	2.1 + 4	8.2 + 5	2.0 + 6	7.4 + 5	2.0 + 5
10^0	sh	3.0 + 3	3.0 + 3	3.0 + 3	3.0 + 3	3.0 + 3	3.0 + 3
	dw	6.3 + 4	1.8 + 3	2.8 + 5	7.4 + 5	1.1 + 5	2.5 + 4
	af	2.0 + 4	6.1 + 3	6.6 + 4	8.2 + 4	1.3 + 4	2.1 + 4
	total	8.6 + 4	1.1 + 4	3.5 + 5	8.2 + 5	1.2 + 5	4.9 + 4
10^{-1}	sh	6.8 + 2	6.8 + 2	6.8 + 2	6.8 + 2	6.8 + 2	6.8 + 2
	dw	2.0 + 3	5.7 + 2	6.8 + 3	9.3 + 3	1.8 + 3	3.3 + 3
	af	5.0 + 3	2.0 + 3	1.5 + 4	7.4 + 3	2.6 + 3	6.7 + 3
	total	7.6 + 3	3.2 + 3	2.2 + 4	1.7 + 4	5.1 + 3	1.1 + 4

t_{H_2O} = 10^2 days

Relative Leach Rate	Pathway	Organ					
		Whole Body	Lung	Total Marrow	Skeleton	LLI	Thyroid
10^5	sh	1.5 + 4	1.5 + 4	1.5 + 4	1.5 + 4	1.5 + 4	1.5 + 4
	dw	1.7 + 5	4.3 + 3	6.3 + 5	1.2 + 6	3.6 + 6	1.1 + 5
	af	9.2 + 3	9.5 + 1	3.9 + 4	9.6 + 4	8.5 + 4	2.8 + 3
	total	1.9 + 5	1.9 + 4	6.8 + 5	1.3 + 6	3.7 + 6	1.3 + 5
10^3	sh	1.4 + 4	1.4 + 4	1.4 + 4	1.4 + 4	1.4 + 4	1.4 + 4
	dw	1.7 + 5	3.8 + 3	6.1 + 5	1.2 + 6	3.4 + 6	1.1 + 5
	af	9.1 + 3	8.4 + 1	3.9 + 4	9.6 + 4	8.1 + 4	2.7 + 3
	total	1.9 + 5	1.8 + 4	6.7 + 5	1.3 + 6	3.5 + 6	1.2 + 5
10^1	sh	1.6 + 3	1.6 + 3	1.6 + 3	1.6 + 3	1.6 + 3	1.6 + 3
	dw	8.9 + 4	4.2 + 2	3.9 + 5	1.0 + 6	5.4 + 5	2.0 + 4
	af	6.8 + 3	9.3 + 0	3.1 + 4	8.3 + 4	1.6 + 4	7.4 + 2
	total	9.8 + 4	2.1 + 3	4.2 + 5	1.1 + 6	5.5 + 5	2.2 + 4
10^0	sh	1.6 + 2	1.6 + 2	1.6 + 2	1.6 + 2	1.6 + 2	1.6 + 2
	dw	2.5 + 4	4.2 + 1	1.2 + 5	3.1 + 5	6.7 + 4	3.0 + 3
	af	2.0 + 3	9.2 - 1	9.4 + 3	2.6 + 4	2.7 + 3	1.6 + 2
	total	2.8 + 4	2.0 + 2	2.3 + 5	3.4 + 5	7.0 + 4	3.3 + 3
10^{-1}	sh	1.6 + 0	1.6 + 0	1.6 + 0	1.6 + 0	1.6 + 0	1.6 + 0
	dw	2.5 + 2	4.2 - 1	1.2 + 3	3.1 + 3	6.7 + 2	3.0 + 1
	af	2.0 + 1	9.2 - 3	9.4 + 1	2.6 + 2	2.7 + 1	1.6 + 0
	total	2.8 + 2	2.0 + 0	2.3 + 3	3.4 + 3	7.0 + 2	3.3 + 1

t_{H_2O} = 10^3 days

Relative Leach Rate	Pathway	Organ					
		Whole Body	Lung	Total Marrow	Skeleton	LLI	Thyroid
10^5	sh	1.7 + 3	1.7 + 3	1.7 + 3	1.7 + 3	1.7 + 3	1.7 + 3
	dw	1.4 + 4	5.0 + 2	3.8 + 4	1.9 + 4	6.0 + 5	1.9 + 4
	af	3.1 + 2	1.1 + 1	8.4 + 2	4.2 + 2	1.3 + 4	4.1 + 2
	total	1.6 + 4	2.2 + 3	4.0 + 4	2.1 + 4	6.2 + 5	2.1 + 4
10^3	sh	1.6 + 3	1.6 + 3	1.6 + 3	1.6 + 3	1.6 + 3	1.6 + 3
	dw	1.3 + 4	4.9 + 2	3.7 + 4	1.8 + 4	5.8 + 5	1.8 + 4
	af	3.0 + 2	1.1 + 1	8.2 + 2	4.0 + 2	1.3 + 4	4.0 + 2
	total	1.5 + 4	2.1 + 3	3.9 + 4	2.0 + 4	5.9 + 5	2.0 + 4
10^1	sh	2.4 + 2	2.4 + 2	2.4 + 2	2.4 + 2	2.4 + 2	2.4 + 2
	dw	2.0 + 3	7.2 + 1	5.5 + 3	2.7 + 3	8.7 + 4	2.7 + 3
	af	4.4 + 1	1.6 + 0	1.2 + 2	6.0 + 1	1.9 + 3	6.0 + 1
	total	2.3 + 3	3.1 + 2	5.8 + 3	3.0 + 3	8.9 + 4	3.0 + 3
10^0	sh	2.3 + 1	2.3 + 1	2.3 + 1	2.3 + 1	2.3 + 1	2.3 + 1
	dw	1.9 + 2	7.1 + 0	5.3 + 2	2.6 + 2	8.5 + 3	2.6 + 2
	af	4.3 + 0	1.6 - 1	1.2 + 1	5.9 + 0	1.9 + 2	5.8 + 0
	total	2.2 + 2	3.0 + 1	5.7 + 2	2.9 + 2	8.7 + 3	2.9 + 2
10^{-1}	sh	2.3 - 1	2.3 - 1	2.3 - 1	2.3 - 1	2.3 - 1	2.3 - 1
	dw	1.9 + 1	7.1 - 2	5.3 + 0	2.6 + 0	8.5 + 1	2.6 + 0
	af	4.3 - 2	1.6 - 3	1.2 - 1	5.9 - 2	1.9 + 0	5.8 - 2
	total	2.2 + 0	3.0 - 1	5.7 + 0	2.9 + 0	8.7 + 1	2.9 + 0

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[a] The numbers in this table and in all succeeding tables possess no more than 1 significant digit.

[b] [illegible] of the elements assumed to be leached most rapidly; relative rates of [illegible] Appendix [illegible]

[c] [illegible] 10^5.

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Table G2.

Resultant Population Doses for the Offshore Region of a Large Lake[a] for Releases from the Melt Debris After a PWR-7 Accident

t_{H_2O} = 1 day

Relative Leach Rate	Pathway	Organ: Whole Body	Lung	Total Marrow	Skeleton	LLI	Thyroid
10^5	sh	3.2 + 5	3.2 + 5	3.2 + 5	3.2 + 5	3.2 + 5	3.2 + 5
	dw	4.4 + 6	7.4 + 4	1.9 + 7	5.0 + 7	1.3 + 7	1.3 + 6
	af	4.0 + 5	1.1 + 5	1.4 + 6	2.4 + 6	3.9 + 5	4.8 + 5
	total	5.1 + 6	5.1 + 5	2.0 + 7	5.3 + 7	1.4 + 7	2.1 + 6
10^3	sh	3.2 + 5	3.2 + 5	3.2 + 5	3.2 + 5	3.2 + 5	3.2 + 5
	dw	4.3 + 6	7.3 + 4	1.9 + 7	5.0 + 7	1.2 + 7	1.3 + 6
	af	4.0 + 5	1.1 + 5	1.4 + 6	2.4 + 6	3.8 + 5	4.8 + 5
	total	5.1 + 6	5.0 + 5	2.0 + 7	5.2 + 7	1.3 + 7	2.1 + 6
10^1	sh	2.0 + 5	2.0 + 5	2.0 + 5	2.0 + 5	2.0 + 5	2.0 + 5
	dw	3.6 + 6	6.0 + 4	1.6 + 7	4.4 + 7	4.1 + 6	7.4 + 5
	af	3.7 + 5	1.0 + 5	1.3 + 6	2.1 + 6	2.6 + 5	4.6 + 5
	total	4.2 + 6	3.7 + 5	1.7 + 7	4.7 + 7	4.5 + 6	1.5 + 6
10^0	sh	9.0 + 4	9.0 + 4	9.0 + 4	9.0 + 4	9.0 + 4	9.0 + 4
	dw	1.5 + 6	4.3 + 4	6.7 + 6	1.8 + 7	1.3 + 6	3.9 + 5
	af	2.4 + 5	7.6 + 4	7.9 + 5	9.9 + 5	1.5 + 5	3.7 + 5
	total	1.8 + 6	2.1 + 5	7.6 + 6	1.9 + 7	1.6 + 6	8.6 + 5
10^{-1}	sh	3.5 + 4	3.5 + 4	3.5 + 4	3.5 + 4	3.5 + 4	3.5 + 4
	dw	5.4 + 4	1.6 + 4	1.8 + 5	2.3 + 5	3.4 + 4	7.9 + 4
	af	7.2 + 4	2.9 + 4	2.1 + 5	1.1 + 5	3.9 + 4	1.4 + 5
	total	1.6 + 5	7.9 + 4	4.2 + 5	3.7 + 5	1.1 + 5	2.6 + 5

t_{H_2O} = 10 days

Relative Leach Rate	Patoway	Organ: Whole Body	Lung	Total Marrow	Skeleton	LLI	Thyroid
10^5	sh	1.2 + 5	1.2 + 5	1.2 + 5	1.2 + 5	1.2 + 5	1.2 + 5
	dw	3.8 + 6	3.4 + 4	1.7 + 7	4.5 + 7	1.2 + 7	5.8 + 5
	af	2.6 + 5	4.3 + 4	1.0 + 6	2.1 + 6	2.8 + 5	1.6 + 5
	total	4.2 + 6	2.0 + 5	1.8 + 7	4.8 + 7	1.3 + 7	8.6 + 5
10^3	sh	1.2 + 5	1.2 + 5	1.2 + 5	1.2 + 5	1.2 + 5	1.2 + 5
	dw	3.8 + 6	3.3 + 4	1.7 + 7	4.5 + 7	1.2 + 7	5.7 + 5
	af	2.6 + 5	4.3 + 4	1.0 + 6	2.1 + 6	2.8 + 5	1.6 + 5
	total	4.2 + 6	2.0 + 5	1.8 + 7	4.7 + 7	1.2 + 7	8.4 + 5
10^1	sh	6.7 + 4	6.7 + 4	6.7 + 4	6.7 + 4	6.7 + 4	6.7 + 4
	dw	3.2 + 6	2.5 + 4	1.5 + 7	4.0 + 7	3.9 + 6	3.2 + 5
	af	2.4 + 5	4.2 + 4	9.2 + 5	1.8 + 6	1.8 + 5	1.5 + 5
	total	3.5 + 6	1.4 + 5	1.5 + 7	4.2 + 7	4.1 + 6	5.3 + 5
10^0	sh	4.6 + 4	4.6 + 4	4.6 + 4	4.6 + 4	4.6 + 4	4.6 + 4
	dw	1.3 + 6	2.1 + 4	6.1 + 6	1.6 + 7	1.2 + 6	1.6 + 5
	af	1.5 + 5	3.8 + 4	5.2 + 5	8.2 + 5	9.5 + 4	1.3 + 5
	total	1.6 + 6	1.1 + 5	6.6 + 6	1.7 + 7	1.3 + 6	3.3 + 5
10^{-1}	sh	1.4 + 4	1.4 + 4	1.4 + 4	1.4 + 4	1.4 + 4	1.4 + 4
	dw	3.0 + 4	6.9 + 3	1.1 + 5	1.9 + 5	2.1 + 4	2.2 + 4
	af	3.1 + 4	1.2 + 4	9.1 + 4	4.9 + 4	1.6 + 4	4.1 + 4
	total	7.5 + 4	3.3 + 4	2.2 + 5	2.5 + 5	5.1 + 4	7.9 + 4

t_{H_2O} = 10^2 days

Relative Leach Rate	Pathway	Organ: Whole Body	Lung	Total Marrow	Skeleton	LLI	Thyroid
10^5	sh	4.0 + 4	4.0 + 4	4.0 + 4	4.0 + 4	4.0 + 4	4.0 + 4
	dw	2.1 + 6	6.4 + 3	9.4 + 6	2.5 + 7	8.9 + 6	3.5 + 5
	af	8.4 + 4	7.2 + 1	3.8 + 5	1.0 + 6	1.5 + 5	7.7 + 3
	total	2.2 + 6	4.7 + 4	9.8 + 6	2.6 + 7	1.0 + 7	3.9 + 5
10^3	sh	3.8 + 4	3.8 + 4	3.8 + 4	3.9 + 4	3.8 + 4	3.8 + 4
	dw	2.1 + 6	6.1 + 3	9.4 + 6	2.5 + 7	8.6 + 6	3.4 + 5
	af	8.4 + 4	6.9 + 1	3.8 + 5	1.0 + 6	1.5 + 5	7.6 + 3
	total	2.2 + 6	4.4 + 4	9.8 + 6	2.6 + 7	8.7 + 6	3.8 + 5
10^1	sh	5.5 + 3	5.5 + 3	5.5 + 3	5.5 + 3	5.5 + 3	5.5 + 3
	dw	1.7 + 6	8.7 + 2	8.0 + 6	2.2 + 7	2.5 + 6	1.4 + 5
	af	7.3 + 4	9.9 + 0	3.4 + 5	9.2 + 5	7.2 + 4	5.0 + 3
	total	1.8 + 6	6.4 + 3	8.4 + 6	2.3 + 7	2.5 + 6	1.4 + 5
10^0	sh	5.5 + 2	5.5 + 2	5.5 + 2	5.5 + 2	5.5 + 2	5.5 + 2
	dw	5.3 + 5	8.7 + 1	2.5 + 6	6.7 + 6	5.4 + 5	3.7 + 4
	af	2.3 + 4	9.9 − 1	1.0 + 5	2.9 + 5	2.0 + 4	1.5 + 3
	total	5.6 + 5	6.4 + 2	2.6 + 6	7.1 + 6	5.6 + 5	3.8 + 4
10^{-1}	sh	5.5 + 0	5.5 + 0	5.5 + 0	5.5 + 0	5.5 + 0	5.5 + 0
	dw	5.2 + 3	8.7 − 1	2.5 + 4	6.7 + 4	5.4 + 3	3.7 + 2
	af	2.3 + 2	9.9 − 3	1.0 + 3	2.9 + 3	2.0 + 2	1.5 + 1
	total	5.5 + 3	6.4 + 0	2.6 + 4	7.0 + 4	5.6 + 3	3.8 + 2

t_{H_2O} = 10^3 days

Relative Leach Rate	Pathway	Organ: Whole Body	Lung	Total Marrow	Skeleton	LLI	Thyroid
10^5	sh	7.2 + 3	7.2 + 3	7.2 + 3	7.2 + 3	7.2 + 3	7.2 + 3
	dw	3.0 + 4	1.1 + 3	8.2 + 4	4.0 + 4	1.3 + 6	4.0 + 4
	af	3.3 + 2	1.2 + 1	9.2 + 2	4.6 + 2	1.5 + 4	4.5 + 2
	total	3.7 + 4	8.3 + 3	8.9 + 4	4.8 + 4	1.3 + 6	4.7 + 4
10^3	sh	7.0 + 3	7.0 + 3	7.0 + 3	7.0 + 3	7.0 + 3	7.0 + 3
	dw	2.9 + 4	1.0 + 3	7.9 + 4	3.9 + 4	1.3 + 6	3.9 + 4
	af	3.2 + 2	1.2 + 1	8.9 + 2	4.4 + 2	1.4 + 4	4.4 + 2
	total	3.6 + 4	8.0 + 3	8.6 + 4	4.6 + 4	1.3 + 6	4.6 + 4
10^1	sh	1.0 + 3	1.0 + 3	1.0 + 3	1.0 + 3	1.0 + 3	1.0 + 3
	dw	4.3 + 3	1.6 + 2	1.2 + 4	5.8 + 3	1.9 + 5	5.8 + 3
	af	4.8 + 1	1.8 + 0	1.3 + 2	6.6 + 1	2.1 + 3	6.5 + 1
	total	5.3 + 3	1.2 + 3	1.3 + 4	7.0 + 3	1.9 + 5	6.9 + 3
10^0	sh	1.0 + 2	1.0 + 2	1.0 + 2	1.0 + 2	1.0 + 2	1.0 + 2
	dw	4.2 + 2	1.5 + 1	1.2 + 3	5.7 + 2	1.8 + 4	5.6 + 2
	af	4.7 + 0	1.7 − 1	1.3 + 1	6.4 + 0	2.1 + 2	6.4 + 0
	total	5.2 + 2	1.2 + 2	1.3 + 3	6.7 + 2	1.8 + 4	6.7 + 2
10^{-1}	sh	1.0 + 0	1.0 + 0	1.0 + 0	1.0 + 0	1.0 + 0	1.0 + 0
	dw	4.2 + 0	1.5 − 1	1.1 + 1	5.7 + 0	1.8 + 2	2.1 − 2
	af	4.7 − 2	1.7 − 3	1.3 − 1	6.4 − 2	2.1 + 0	6.4 + 2
	total	5.2 + 0	1.2 + 0	1.2 + 1	6.7 + 0	1.8 + 2	2.1 + 2

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[a] The offshore region of a large lake is taken to be all of the lake except for the nearshore region.

Table G3.

Resultant Population Doses for a Small Estuary for Releases from the Melt Debris After a PWR-7 Accident

t_{H_2O} = 1 day

Relative Leach Rate	Pathway	Whole Body	Lung	Total Marrow	Skeleton	LLI	Thyroid
10^5	sh	1.1 + 7	1.1 + 7	1.1 + 7	1.1 + 7	1.1 + 7	1.1 + 7
	af	1.3 + 7	2.0 + 6	3.7 + 7	4.4 + 7	3.4 + 8	2.2 + 7
	total	2.4 + 7	1.3 + 7	4.8 + 7	5.5 + 7	3.5 + 8	3.3 + 7
10^3	sh	1.0 + 7	1.0 + 7	1.0 + 7	1.0 + 7	1.0 + 7	1.0 + 7
	af	1.3 + 7	2.0 + 6	3.7 + 7	4.4 + 7	3.2 + 8	2.2 + 7
	total	2.3 + 7	1.2 + 7	4.7 + 7	5.4 + 7	3.3 + 8	3.2 + 7
10^1	sh	4.6 + 6	4.6 + 6	4.6 + 6	4.6 + 6	4.6 + 6	4.6 + 6
	af	4.3 + 6	1.3 + 6	1.2 + 7	1.8 + 7	5.7 + 7	5.0 + 6
	total	8.9 + 6	5.9 + 6	1.7 + 7	2.3 + 7	6.2 + 7	1.0 + 7
10^0	sh	1.9 + 6	1.9 + 6	1.9 + 6	1.9 + 6	1.9 + 6	1.9 + 6
	af	6.0 + 5	2.0 + 5	1.7 + 6	1.7 + 6	6.4 + 6	8.4 + 5
	total	2.5 + 6	2.1 + 6	3.6 + 6	3.6 + 6	8.3 + 6	2.7 + 6
10^{-1}	sh	6.7 + 5	6.7 + 5	6.7 + 5	6.7 + 5	6.7 + 5	6.7 + 5
	af	2.0 + 4	2.0 + 4	1.8 + 5	1.1 + 5	9.0 + 4	1.5 + 5
	total	6.9 + 5	6.9 + 5	8.6 + 5	7.8 + 5	7.6 + 5	8.1 + 5

t_{H_2O} = 10 days

Relative Leach Rate	Pathway	Whole Body	Lung	Total Marrow	Skeleton	LLI	Thyroid
10^5	sh	1.0 + 7	1.0 + 7	1.0 + 7	1.0 + 7	1.0 + 7	1.0 + 7
	af	8.0 + 6	1.1 + 6	2.1 + 7	1.3 + 7	3.1 + 8	1.0 + 7
	total	1.7 + 7	1.0 + 7	3.1 + 7	2.3 + 7	3.2 + 8	2.0 + 7
10^3	sh	8.5 + 6	8.5 + 6	8.5 + 6	8.5 + 6	8.5 + 6	8.5 + 6
	af	7.5 + 6	1.0 + 6	2.1 + 7	1.3 + 7	2.9 + 8	1.0 + 7
	total	1.6 + 7	9.5 + 6	3.0 + 7	2.2 + 7	3.0 + 8	1.9 + 7
10^1	sh	2.3 + 6	2.3 + 6	2.3 + 6	2.3 + 6	2.3 + 6	2.3 + 6
	af	1.8 + 6	4.0 + 5	4.9 + 6	4.4 + 6	4.5 + 7	2.1 + 6
	total	4.1 + 6	2.7 + 6	7.2 + 6	6.7 + 6	4.7 + 7	4.4 + 6
10^0	sh	1.0 + 6	1.0 + 6	1.0 + 6	1.0 + 6	1.0 + 6	1.0 + 6
	af	3.0 + 5	8.0 + 4	1.0 + 6	1.3 + 6	4.7 + 6	3.4 + 5
	total	1.3 + 6	1.1 + 6	2.0 + 6	2.3 + 6	5.7 + 6	1.3 + 6
10^{-1}	sh	2.8 + 5	2.8 + 5	2.8 + 5	2.8 + 5	2.8 + 5	2.8 + 5
	af	3.0 + 4	1.1 + 4	9.0 + 4	5.7 + 4	6.0 + 4	7.0 + 4
	total	3.0 + 5	2.9 + 5	3.6 + 5	3.3 + 5	3.4 + 5	3.5 + 5

t_{H_2O} = 10^2 days

Relative Leach Rate	Pathway	Whole Body	Lung	Total Marrow	Skeleton	LLI	Thyroid
10^5	sh	5.7 + 6	5.7 + 6	5.7 + 6	5.7 + 6	5.7 + 6	5.7 + 6
	af	5.4 + 6	2.1 + 5	1.6 + 7	8.9 + 6	2.3 + 8	7.1 + 6
	total	1.1 + 7	5.9 + 6	2.1 + 7	1.5 + 7	2.4 + 8	1.3 + 7
10^3	sh	5.5 + 6	5.5 + 6	5.5 + 6	5.5 + 6	5.5 + 6	5.5 + 6
	af	5.2 + 6	2.0 + 5	1.5 + 7	8.7 + 6	2.2 + 8	6.9 + 6
	total	1.1 + 7	5.7 + 6	2.1 + 7	1.4 + 7	2.3 + 8	1.3 + 7
10^1	sh	7.9 + 5	7.9 + 5	7.9 + 5	7.9 + 5	7.9 + 5	7.9 + 5
	af	8.7 + 5	2.8 + 4	2.6 + 6	2.5 + 6	3.3 + 7	1.1 + 6
	total	1.7 + 6	8.2 + 5	3.4 + 6	3.2 + 6	3.4 + 7	1.8 + 6
10^0	sh	7.9 + 4	7.9 + 4	7.9 + 4	7.9 + 4	7.9 + 4	7.9 + 4
	af	1.1 + 5	2.8 + 3	3.6 + 5	5.3 + 5	3.3 + 6	1.1 + 5
	total	1.8 + 5	8.2 + 4	4.4 + 5	6.1 + 5	3.4 + 6	1.8 + 5
10^{-1}	sh	7.9 + 2	7.9 + 2	7.9 + 2	7.9 + 2	7.9 + 2	7.9 + 2
	af	1.1 + 3	2.8 + 1	3.6 + 3	5.3 + 5	3.3 + 4	1.1 + 3
	total	1.8 + 3	8.2 + 2	4.4 + 3	6.1 + 3	3.4 + 4	1.8 + 3

t_{H_2O} = 10^3 days

Relative Leach Rate	Pathway	Whole Body	Lung	Total Marrow	Skeleton	LLI	Thyroid
10^5	sh	6.3 + 5	6.3 + 5	6.3 + 5	6.3 + 5	6.3 + 5	6.3 + 5
	af	1.0 + 6	3.5 + 4	2.7 + 6	1.4 + 6	4.2 + 7	1.3 + 6
	total	1.6 + 6	6.7 + 5	3.3 + 6	2.0 + 6	4.3 + 7	1.9 + 6
10^3	sh	6.1 + 5	6.1 + 5	6.1 + 5	6.1 + 5	6.1 + 5	6.1 + 5
	af	9.3 + 5	3.5 + 4	2.6 + 6	1.3 + 6	4.1 + 7	1.3 + 6
	total	1.5 + 6	6.5 + 5	3.2 + 6	1.9 + 6	4.2 + 7	1.9 + 6
10^1	sh	9.1 + 4	9.1 + 4	9.1 + 4	9.1 + 4	9.1 + 4	9.1 + 4
	af	1.4 + 5	5.2 + 3	3.8 + 5	1.9 + 5	6.1 + 6	1.9 + 5
	total	2.3 + 5	9.6 + 4	4.7 + 5	2.8 + 5	6.2 + 6	2.8 + 5
10^0	sh	8.9 + 3	8.9 + 3	8.9 + 3	8.9 + 3	8.9 + 3	8.9 + 3
	af	1.4 + 4	5.0 + 2	3.7 + 4	1.9 + 4	6.0 + 5	1.8 + 4
	total	2.3 + 4	9.4 + 3	4.6 + 4	2.7 + 4	6.1 + 5	2.7 + 4
10^{-1}	sh	8.9 + 1	8.9 + 1	8.9 + 1	8.9 + 1	8.9 + 1	8.9 + 1
	af	1.4 + 2	4.0 + 0	3.7 + 2	1.9 + 2	6.0 + 3	1.8 + 2
	total	2.3 + 2	9.3 + 1	4.6 + 2	2.7 + 2	6.1 + 3	2.7 + 2

Table G4.

Resultant Population Doses for a Large Estuary for Releases from the Melt Debris After a PWR-7 Accident

t_{H_2O} = 1 day

Relative Leach Rate	Pathway	Organ: Whole Body	Lung	Total Marrow	Skeleton	LLI	Thyroid
10^5	sh	2.6 + 6	2.6 + 6	2.6 + 6	2.6 + 6	2.6 + 6	2.6 + 6
	af	1.3 + 7	2.4 + 6	4.0 + 7	6.0 + 7	2.9 + 8	1.9 + 7
	total	1.6 + 7	5.0 + 6	4.3 + 7	6.3 + 7	2.9 + 8	2.2 + 7
10^3	sh	2.5 + 6	2.5 + 6	2.5 + 6	2.5 + 6	2.5 + 6	2.5 + 6
	af	1.3 + 7	2.4 + 6	3.9 + 7	5.9 + 7	2.8 + 8	1.9 + 7
	total	1.6 + 7	4.9 + 6	4.2 + 7	6.2 + 7	2.8 + 8	2.2 + 7
10^1	sh	1.3 + 6	1.3 + 6	1.3 + 6	1.3 + 6	1.3 + 6	1.3 + 6
	af	4.9 + 6	1.2 + 6	1.6 + 7	2.7 + 7	5.2 + 7	8.4 + 6
	total	6.2 + 6	2.5 + 6	1.7 + 7	2.8 + 7	5.3 + 7	1.0 + 7
10^0	sh	5.2 + 5	5.2 + 5	5.2 + 5	5.2 + 5	5.2 + 5	5.2 + 5
	af	5.8 + 5	2.3 + 5	1.7 + 6	2.0 + 6	5.9 + 6	9.1 + 5
	total	1.1 + 6	7.5 + 5	2.2 + 6	2.5 + 6	6.4 + 6	1.4 + 6
10^{-1}	sh	1.9 + 5	1.9 + 5	1.9 + 5	1.9 + 5	1.9 + 5	1.9 + 5
	af	6.0 + 4	2.0 + 4	2.1 + 5	1.1 + 5	9.0 + 4	1.4 + 5
	total	2.5 + 5	2.1 + 5	4.0 + 5	3.0 + 5	2.8 + 5	3.3 + 5

t_{H_2O} = 10 days

Relative Leach Rate	Pathway	Organ: Whole Body	Lung	Total Marrow	Skeleton	LLI	Thyroid
10^5	sh	1.8 + 6	1.8 + 6	1.8 + 6	1.8 + 6	1.8 + 6	1.8 + 6
	af	7.2 + 6	1.0 + 6	2.0 + 7	1.3 + 7	2.6 + 8	9.0 + 6
	total	9.0 + 6	2.8 + 6	2.2 + 7	1.5 + 7	2.6 + 8	1.1 + 7
10^3	sh	1.8 + 6	1.8 + 6	1.8 + 6	1.8 + 6	1.8 + 6	1.8 + 6
	af	6.5 + 6	9.0 + 5	1.9 + 7	1.3 + 7	2.5 + 8	9.0 + 6
	total	8.7 + 6	2.7 + 6	2.1 + 7	1.5 + 7	2.5 + 8	1.1 + 7
10^1	sh	5.6 + 5	5.6 + 5	5.6 + 5	5.6 + 5	5.6 + 5	5.6 + 5
	af	1.6 + 6	3.9 + 5	4.6 + 6	4.3 + 6	4.0 + 7	5.8 + 6
	total	2.2 + 6	9.5 + 5	5.2 + 6	4.9 + 6	4.0 + 7	6.4 + 6
10^0	sh	2.7 + 5	2.7 + 5	2.7 + 5	2.7 + 5	2.7 + 5	2.7 + 5
	af	3.2 + 5	7.0 + 4	1.0 + 6	1.3 + 6	4.2 + 6	3.1 + 5
	total	5.9 + 5	3.4 + 5	1.3 + 6	1.6 + 6	4.5 + 6	5.8 + 5
10^{-1}	sh	7.8 + 4	7.8 + 4	7.8 + 4	7.8 + 4	7.8 + 4	7.8 + 4
	af	3.0 + 4	1.3 + 4	9.0 + 4	5.0 + 4	5.0 + 4	5.0 + 4
	total	1.1 + 5	9.1 + 4	1.7 + 5	1.2 + 5	1.2 + 5	1.2 + 5

t_{H_2O} = 10^2 days

Relative Leach Rate	Pathway	Organ: Whole Body	Lung	Total Marrow	Skeleton	LLI	Thyroid
10^5	sh	1.0 + 6	1.0 + 6	1.0 + 6	1.0 + 6	1.0 + 6	1.0 + 6
	af	4.8 + 6	2.0 + 5	1.3 + 7	7.9 + 6	2.0 + 8	6.3 + 6
	total	5.8 + 6	1.2 + 6	1.4 + 7	8.9 + 6	2.0 + 8	7.3 + 6
10^3	sh	9.8 + 5	9.8 + 5	9.8 + 5	9.8 + 5	9.8 + 5	9.8 + 5
	af	4.6 + 6	1.8 + 5	1.3 + 7	7.7 + 6	2.0 + 8	6.0 + 6
	total	5.6 + 6	1.2 + 6	1.4 + 7	8.7 + 6	2.0 + 8	7.0 + 6
10^1	sh	1.4 + 5	1.4 + 5	1.4 + 5	1.4 + 5	1.4 + 5	1.4 + 5
	af	7.7 + 5	2.0 + 4	2.4 + 6	2.4 + 6	2.9 + 7	8.6 + 5
	total	9.1 + 5	1.6 + 5	2.5 + 6	2.5 + 6	2.9 + 7	1.0 + 6
10^0	sh	1.4 + 4	1.4 + 4	1.4 + 4	1.4 + 4	1.4 + 4	1.4 + 4
	af	1.0 + 5	2.0 + 3	3.3 + 5	5.2 + 5	2.9 + 6	8.6 + 4
	total	1.1 + 5	1.6 + 4	3.5 + 5	5.3 + 5	2.9 + 6	1.0 + 5
10^{-1}	sh	1.4 + 2	1.4 + 2	1.4 + 2	1.4 + 2	1.4 + 2	1.4 + 2
	af	1.0 + 3	2.0 + 1	3.3 + 3	5.2 + 3	2.9 + 4	8.6 + 2
	total	1.1 + 3	1.6 + 2	3.5 + 3	5.3 + 3	2.9 + 4	1.0 + 3

t_{H_2O} = 10^3 days

Relative Leach Rate	Pathway	Organ: Whole Body	Lung	Total Marrow	Skeleton	LLI	Thyroid
10^5	sh	1.4 + 5	1.4 + 5	1.4 + 5	1.4 + 5	1.4 + 5	1.4 + 5
	af	8.3 + 5	4.0 + 4	2.4 + 6	1.2 + 6	3.7 + 7	1.2 + 6
	total	9.7 + 5	1.8 + 5	2.5 + 6	1.3 + 6	3.7 + 7	1.3 + 6
10^3	sh	1.3 + 5	1.3 + 5	1.3 + 5	1.3 + 5	1.3 + 5	1.3 + 5
	af	8.1 + 5	4.0 + 4	2.3 + 6	1.2 + 6	3.6 + 7	1.2 + 6
	total	9.4 + 5	1.7 + 5	2.4 + 6	1.3 + 6	3.6 + 7	1.3 + 6
10^1	sh	2.0 + 4	2.0 + 4	2.0 + 4	2.0 + 4	2.0 + 4	2.0 + 4
	af	1.2 + 5	4.0 + 3	3.3 + 5	1.7 + 5	5.3 + 6	1.6 + 5
	total	1.4 + 5	2.4 + 4	3.5 + 5	1.9 + 5	5.3 + 6	1.8 + 5
10^0	sh	1.9 + 3	1.9 + 3	1.9 + 3	1.9 + 3	1.9 + 3	1.9 + 3
	af	1.2 + 4	5.0 + 2	3.3 + 4	1.7 + 4	5.3 + 5	1.6 + 4
	total	1.4 + 4	2.4 + 3	3.5 + 4	1.9 + 4	5.3 + 5	1.8 + 4
10^{-1}	sh	1.9 + 1	1.9 + 1	1.9 + 1	1.9 + 1	1.9 + 1	1.9 + 1
	af	1.2 + 2	5.0 + 0	3.3 + 2	1.7 + 2	5.3 + 3	1.6 + 2
	total	1.4 + 2	2.4 + 1	3.5 + 2	1.9 + 2	5.3 + 3	1.8 + 2

Table G5.

Resultant Population Doses for the Nearshore Region of an Ocean for Releases from the Melt Debris after a PWR-7 Accident[a]

t_{H_2O} = 1 day

Relative Leach Rate	Pathway	Organ: Whole Body	Lung	Total Marrow	Skeleton	LLI	Thyroid
10^5	sh	6.0 + 4	6.0 + 4	6.0 + 4	6.0 + 4	6.0 + 4	6.0 + 4
	af	2.2 + 6	2.9 + 5	6.4 + 6	1.0 + 7	6.7 + 7	3.1 + 6
	total	2.2 + 6	3.5 + 5	6.4 + 6	1.0 + 7	6.7 + 7	3.1 + 6
10^3	sh	4.7 + 4	4.7 + 4	4.7 + 4	4.7 + 4	4.7 + 4	4.7 + 4
	af	2.0 + 6	2.8 + 5	6.2 + 6	1.0 + 7	6.3 + 7	3.1 + 6
	total	2.0 + 6	3.3 + 5	6.2 + 6	1.0 + 7	6.3 + 7	3.1 + 6
10^1	sh	5.3 + 3	5.3 + 3	5.3 + 3	5.3 + 3	5.3 + 3	5.3 + 3
	af	5.8 + 5	1.5 + 5	1.7 + 6	2.7 + 6	6.9 + 6	6.5 + 5
	total	5.8 + 5	1.5 + 5	1.7 + 6	2.7 + 6	6.9 + 6	6.5 + 5
10^0	sh	1.1 + 3	1.1 + 3	1.1 + 3	1.1 + 3	1.1 + 3	1.1 + 3
	af	9.0 + 4	3.4 + 4	2.6 + 5	3.3 + 5	7.7 + 5	1.3 + 5
	total	9.0 + 4	3.4 + 4	2.6 + 5	3.3 + 5	7.7 + 5	1.3 + 5
10^{-1}	sh	2.0 + 2	2.0 + 2	2.0 + 2	2.0 + 2	2.0 + 2	2.0 + 2
	af	1.4 + 4	5.5 + 3	4.0 + 4	2.0 + 4	1.5 + 4	2.6 + 4
	total	1.4 + 4	5.7 + 3	4.0 + 4	2.0 + 4	1.5 + 4	2.6 + 4

t_{H_2O} = 10 days

Relative Leach Rate	Pathway	Organ: Whole Body	Lung	Total Marrow	Skeleton	LLI	Thyroid
10^5	sh	4.8 + 4	4.8 + 4	4.8 + 4	4.8 + 4	4.8 + 4	4.8 + 4
	af	1.5 + 6	1.5 + 5	3.9 + 6	2.0 + 6	5.7 + 7	1.6 + 6
	total	1.6 + 6	2.7 + 5	4.0 + 6	2.1 + 6	5.7 + 7	1.7 + 6
10^3	sh	3.8 + 4	3.8 + 4	3.8 + 4	3.8 + 4	3.8 + 4	3.8 + 4
	af	1.4 + 6	1.4 + 5	3.7 + 6	2.1 + 6	5.4 + 7	1.6 + 6
	total	1.5 + 6	2.4 + 5	3.8 + 6	2.2 + 6	5.4 + 7	1.7 + 6
10^1	sh	8.2 + 3	8.2 + 3	8.2 + 3	8.2 + 3	8.2 + 3	8.2 + 3
	af	2.6 + 5	5.0 + 4	7.5 + 5	6.3 + 5	7.8 + 6	2.8 + 5
	total	2.7 + 5	5.8 + 4	7.6 + 5	6.4 + 5	7.8 + 6	2.9 + 5
10^0	sh	8.7 + 2	8.7 + 2	8.7 + 2	8.7 + 2	8.7 + 2	8.7 + 2
	af	5.0 + 4	1.2 + 4	1.6 + 5	1.9 + 5	8.1 + 5	5.0 + 4
	total	5.1 + 4	1.3 + 4	1.6 + 5	1.9 + 5	8.1 + 5	5.1 + 4
10^{-1}	sh	2.5 + 1	2.5 + 1	2.5 + 1	2.5 + 1	2.5 + 1	2.5 + 1
	af	5.8 + 3	2.3 + 3	1.7 + 4	9.2 + 3	1.1 + 4	7.7 + 3
	total	5.8 + 3	2.3 + 3	1.7 + 4	9.2 + 3	1.1 + 4	7.7 + 3

t_{H_2O} = 10^2 days

Relative Leach Rate	Pathway	Organ: Whole Body	Lung	Total Marrow	Skeleton	LLI	Thyroid
10^5	sh	2.5 + 3	2.5 + 3	2.5 + 3	2.5 + 3	2.5 + 3	2.5 + 3
	af	6.3 + 5	3.0 + 4	1.7 + 6	1.0 + 6	2.7 + 7	8.0 + 5
	total	6.3 + 5	3.2 + 4	1.7 + 6	1.0 + 6	2.7 + 7	8.0 + 5
10^3	sh	2.1 + 3	2.1 + 3	2.1 + 3	2.1 + 3	2.1 + 3	2.1 + 3
	af	6.0 + 5	2.7 + 4	1.7 + 6	9.9 + 5	2.6 + 6	7.7 + 5
	total	6.0 + 5	2.9 + 4	1.7 + 6	9.9 + 5	2.6 + 7	7.7 + 5
10^1	sh	1.8 + 2	1.8 + 2	1.8 + 2	1.8 + 2	1.8 + 2	1.8 + 2
	af	9.8 + 4	3.1 + 3	3.0 + 5	3.0 + 5	3.7 + 6	1.1 + 5
	total	9.8 + 4	3.3 + 3	3.0 + 5	3.0 + 5	3.7 + 6	1.1 + 5
10^0	sh	1.8 + 1	1.8 + 1	1.8 + 1	1.8 + 1	1.8 + 1	1.8 + 1
	af	1.3 + 4	3.1 + 2	4.3 + 4	6.8 + 4	3.7 + 5	1.2 + 4
	total	1.3 + 4	3.3 + 2	4.3 + 4	6.8 + 4	3.7 + 5	1.2 + 4
10^{-1}	sh	1.8 − 1	1.8 − 1	1.8 − 1	1.8 − 1	1.8 − 1	1.8 − 1
	af	1.3 + 2	3.1 + 0	4.3 + 2	6.8 + 2	3.7 + 3	1.2 + 2
	total	1.3 + 2	3.3 + 0	4.3 + 2	6.8 + 2	3.7 + 3	1.2 + 2

t_{H_2O} = 10^3 days

Relative Leach Rate	Pathway	Organ: Whole Body	Lung	Total Marrow	Skeleton	LLI	Thyroid
10^5	sh	7.7 + 2	7.7 + 2	7.7 + 2	7.7 + 2	7.7 + 2	7.7 + 2
	af	1.1 + 5	3.8 + 3	2.9 + 5	1.4 + 5	4.6 + 6	1.4 + 5
	total	1.1 + 5	4.6 + 3	2.9 + 5	1.4 + 5	4.6 + 6	1.4 + 5
10^3	sh	7.4 + 2	7.4 + 2	7.4 + 2	7.4 + 2	7.4 + 2	7.4 + 2
	af	1.0 + 5	3.7 + 3	2.8 + 5	1.4 + 5	4.5 + 6	1.4 + 5
	total	1.0 + 5	4.4 + 3	2.8 + 5	1.4 + 5	4.5 + 6	1.4 + 5
10^1	sh	1.1 + 2	1.1 + 2	1.1 + 2	1.1 + 2	1.1 + 2	1.1 + 2
	af	1.5 + 4	5.6 + 2	4.2 + 4	2.1 + 4	6.7 + 5	2.1 + 4
	total	1.5 + 4	6.7 + 2	4.2 + 4	2.1 + 4	6.7 + 5	2.1 + 4
10^0	sh	1.1 + 1	1.1 + 1	1.1 + 1	1.1 + 1	1.1 + 1	1.1 + 1
	af	1.5 + 3	5.6 + 1	4.2 + 3	2.1 + 3	6.7 + 4	2.1 + 3
	total	1.5 + 3	6.7 + 1	4.2 + 3	2.1 + 3	6.7 + 4	2.1 + 3
10^{-1}	sh	1.1 − 1	1.1 − 1	1.1 − 1	1.1 − 1	1.1 − 1	1.1 − 1
	af	1.5 + 1	5.6 − 1	4.2 + 3	2.1 + 1	6.7 + 2	2.1 + 1
	total	1.5 + 1	6.7 − 1	4.2 + 1	2.1 + 1	6.7 + 2	2.1 + 1

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[a] The estimated doses for some organs for 1-day groundwater travel times could be significantly higher than those indicated if the effects of Am-251 are included.

Table G6.

Resultant Population Doses for the Offshore Region of an Ocean for Releases from the Melt Debris after a PWR-7 Accident[a]

t_{H_2O} = 1 day

Relative Leach Rate	Pathway	Organ: Whole Body	Lung	Total Marrow	Skeleton	LLI	Thyroid
10^5	sh	3.9 + 3	3.9 + 3	3.9 + 3	3.9 + 3	3.9 + 3	3.9 + 3
	af	2.9 + 5	2.5 + 4	9.9 + 5	2.1 + 6	3.9 + 6	2.9 + 5
	total	2.9 + 5	2.9 + 4	9.9 + 5	2.1 + 6	3.9 + 6	2.9 + 5
10^3	sh	3.5 + 3	3.5 + 3	3.5 + 3	3.5 + 3	3.5 + 3	3.5 + 3
	af	2.7 + 5	2.4 + 4	9.6 + 5	2.1 + 6	3.9 + 6	2.8 + 5
	total	2.7 + 5	2.8 + 4	9.6 + 5	2.1 + 6	3.9 + 6	2.8 + 5
10^1	sh	8.3 + 2	8.3 + 2	8.3 + 2	8.3 + 2	8.3 + 2	8.3 + 2
	af	8.4 + 4	1.6 + 4	2.8 + 5	6.0 + 5	6.5 + 5	7.2 + 4
	total	8.4 + 4	1.6 + 4	2.8 + 5	6.0 + 5	6.5 + 5	7.2 + 4
10^0	sh	2.0 + 2	2.0 + 2	2.0 + 2	2.0 + 2	2.0 + 2	2.0 + 2
	af	1.6 + 4	4.9 + 3	4.9 + 4	6.5 + 4	7.6 + 4	2.2 + 4
	total	1.6 + 4	5.1 + 3	4.9 + 4	6.5 + 4	7.6 + 4	2.2 + 4
10^{-1}	sh	4.9 + 1	4.9 + 1	4.9 + 1	4.9 + 1	4.9 + 1	4.9 + 1
	af	3.1 + 3	1.3 + 3	1.1 + 4	4.7 + 3	2.4 + 3	6.1 + 3
	total	3.1 + 3	1.3 + 3	1.1 + 4	4.7 + 3	2.4 + 3	6.1 + 3

t_{H_2O} = 10 days

Relative Leach Rate	Pathway	Organ: Whole Body	Lung	Total Marrow	Skeleton	LLI	Thyroid
10^5	sh	3.2 + 3	3.2 + 3	3.2 + 3	3.2 + 3	3.2 + 3	3.2 + 3
	af	1.2 + 5	1.1 + 4	3.3 + 5	3.5 + 5	3.6 + 6	1.4 + 5
	total	1.2 + 5	1.4 + 4	3.3 + 5	3.5 + 5	3.6 + 6	1.4 + 5
10^3	sh	2.8 + 3	2.8 + 3	2.8 + 3	2.8 + 3	2.8 + 3	2.8 + 3
	af	1.1 + 5	1.0 + 4	3.2 + 5	3.4 + 5	3.5 + 6	1.3 + 5
	total	1.1 + 5	1.3 + 4	3.2 + 5	3.4 + 5	3.5 + 6	1.3 + 5
10^1	sh	4.2 + 2	4.2 + 2	4.2 + 2	4.2 + 2	4.2 + 2	4.2 + 2
	af	2.9 + 4	4.7 + 3	9.2 + 4	1.4 + 5	5.3 + 5	2.9 + 4
	total	2.9 + 4	5.1 + 3	9.2 + 4	1.4 + 5	5.3 + 5	2.9 + 4
10^0	sh	1.0 + 2	1.0 + 2	1.0 + 2	1.0 + 2	1.0 + 2	1.0 + 2
	af	8.8 + 3	1.9 + 3	3.1 + 4	4.9 + 4	5.8 + 4	7.8 + 3
	total	8.9 + 3	2.0 + 3	3.1 + 4	4.9 + 4	5.8 + 4	7.9 + 3
10^{-1}	sh	2.2 + 1	2.2 + 1	2.2 + 1	2.2 + 1	2.2 + 1	2.2 + 1
	af	1.3 + 3	5.1 + 2	3.8 + 3	2.2 + 3	1.2 + 3	1.8 + 3
	total	1.3 + 3	5.3 + 2	3.8 + 3	2.2 + 3	1.2 + 3	1.8 + 3

t_{H_2O} = 10^2 days

Relative Leach Rate	Pathway	Organ: Whole Body	Lung	Total Marrow	Skeleton	LLI	Thyroid
10^5	sh	1.6 + 3	1.6 + 3	1.6 + 3	1.6 + 3	1.6 + 3	1.6 + 3
	af	6.9 + 4	2.6 + 3	2.0 + 5	1.5 + 5	2.8 + 6	8.8 + 4
	total	7.1 + 4	4.2 + 3	2.0 + 5	1.5 + 5	2.8 + 6	8.9 + 4
10^3	sh	1.5 + 3	1.5 + 3	1.5 + 3	1.5 + 3	1.5 + 3	1.5 + 3
	af	6.7 + 4	2.4 + 3	1.9 + 5	1.5 + 5	2.7 + 6	8.4 + 4
	total	6.9 + 4	3.9 + 3	1.9 + 5	1.5 + 5	2.7 + 6	8.6 + 4
10^1	sh	2.0 + 2	2.0 + 2	2.0 + 2	2.0 + 2	2.0 + 2	2.0 + 2
	af	1.4 + 4	3.4 + 2	4.5 + 4	6.7 + 4	4.1 + 5	1.3 + 4
	total	1.4 + 4	5.4 + 2	4.5 + 4	6.7 + 4	4.1 + 5	1.3 + 4
10^0	sh	2.0 + 1	2.0 + 1	2.0 + 1	2.0 + 1	2.0 + 1	2.0 + 1
	af	2.3 + 3	3.4 + 1	8.6 + 3	1.8 + 4	4.1 + 4	1.4 + 3
	total	2.3 + 3	5.4 + 1	8.6 + 3	1.8 + 4	4.1 + 4	1.4 + 3
10^{-1}	sh	2.0 - 1	2.0 - 1	2.0 - 1	2.0 - 1	2.0 - 1	2.0 - 1
	af	2.3 + 1	3.4 + 1	8.6 + 1	1.8 + 2	4.1 + 2	1.4 + 1
	total	2.3 + 1	5.4 + 1	8.6 + 1	1.8 + 2	4.1 + 2	1.4 + 1

t_{H_2O} = 10^3 days

Relative Leach Rate	Pathway	Organ: Whole Body	Lung	Total Marrow	Skeleton	LLI	Thyroid
10^5	sh	2.4 + 2	2.4 + 2	2.4 + 2	2.4 + 2	2.4 + 2	2.4 + 2
	af	1.2 + 4	1.7 + 2	3.1 + 4	1.5 + 4	4.9 + 5	1.5 + 4
	total	1.2 + 4	4.1 + 2	3.1 + 4	1.5 + 4	4.9 + 5	1.5 + 4
10^3	sh	2.4 + 2	2.4 + 2	2.4 + 2	2.4 + 2	2.4 + 2	2.4 + 2
	af	1.1 + 4	1.6 + 2	3.0 + 4	1.5 + 4	4.8 + 5	1.5 + 4
	total	1.1 + 4	4.0 + 2	3.0 + 4	1.5 + 4	4.8 + 5	1.5 + 4
10^1	sh	3.5 + 1	3.5 + 1	3.5 + 1	3.5 + 1	3.5 + 1	3.5 + 1
	af	1.7 + 3	6.5 + 1	4.5 + 3	2.2 + 3	7.1 + 4	2.2 + 3
	total	1.7 + 3	1.0 + 2	4.5 + 3	2.2 + 3	7.1 + 4	2.2 + 3
10^0	sh	3.4 + 0	3.4 + 0	3.4 + 0	3.4 + 0	3.4 + 0	3.4 + 0
	af	1.6 + 2	6.6 + 0	4.2 + 2	2.2 + 2	7.0 + 3	2.1 + 2
	total	1.6 + 2	1.0 + 1	4.2 + 2	2.2 + 2	7.0 + 3	2.1 + 2
10^{-1}	sh	3.4 - 2	3.4 - 2	3.4 - 2	3.4 - 2	3.4 - 2	3.4 - 2
	af	1.6 + 0	6.5 - 2	4.4 + 0	2.1 + 0	7.0 + 1	2.1 + 0
	total	1.6 + 0	9.9 - 2	4.4 + 0	2.1 + 0	7.0 + 1	2.1 + 0

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[a] The estimated doses for some organs for a 1-day groundwater travel time could be significantly higher than those indicated if the effects of Am-251 are included.

Table G7.

Resultant Population Doses for a Dammed River System for Releases from the Melt Debris after a PWR-7 Accident

t_{H_2O} = 1 day

Relative Leach Rate	Pathway	Organ: Whole Body	Lung	Total Marrow	Skeleton	LLI	Thyroid
10^5	sh	8.5 + 5	8.5 + 5	8.5 + 5	8.5 + 5	8.5 + 5	8.5 + 5
	dw	1.8 + 7	7.5 + 5	6.7 + 7	1.4 + 8	3.6 + 8	2.4 + 7
	af	4.1 + 6	1.2 + 6	1.4 + 7	2.3 + 7	1.9 + 7	4.8 + 6
	total	2.3 + 7	2.8 + 6	8.1 + 8	1.6 + 8	3.8 + 8	2.9 + 7
10^3	sh	7.8 + 5	7.8 + 5	7.8 + 5	7.8 + 5	7.8 + 5	7.8 + 5
	dw	1.7 + 7	6.7 + 5	6.5 + 7	1.4 + 8	3.3 + 8	2.4 + 7
	af	4.1 + 6	1.2 + 6	1.4 + 7	2.3 + 7	1.8 + 7	4.7 + 6
	total	2.2 + 7	2.6 + 6	7.9 + 7	1.6 + 8	3.5 + 8	2.9 + 7
10^1	sh	1.9 + 5	1.9 + 5	1.9 + 5	1.9 + 5	1.9 + 5	1.9 + 5
	dw	9.8 + 6	2.5 + 5	4.3 + 7	1.1 + 8	4.8 + 7	5.5 + 6
	af	3.4 + 6	1.1 + 6	1.2 + 7	2.0 + 7	4.1 + 6	3.8 + 6
	total	1.3 + 7	1.5 + 6	5.4 + 7	1.3 + 8	5.2 + 7	9.5 + 6
10^0	sh	5.2 + 4	5.2 + 4	5.2 + 4	5.2 + 4	5.2 + 4	5.2 + 4
	dw	3.8 + 6	1.2 + 5	1.7 + 7	4.5 + 7	7.0 + 6	2.7 + 6
	af	1.9 + 6	6.2 + 5	6.4 + 6	9.0 + 6	1.4 + 6	2.6 + 6
	total	5.8 + 6	8.0 + 5	2.3 + 7	5.4 + 7	8.5 + 6	5.3 + 6
10^{-1}	sh	1.5 + 4	1.5 + 4	1.5 + 4	1.5 + 4	1.5 + 4	1.5 + 4
	dw	1.2 + 5	3.8 + 4	4.2 + 5	5.6 + 5	1.2 + 5	1.9 + 5
	af	4.7 + 5	2.0 + 5	1.4 + 6	7.1 + 5	2.6 + 5	9.0 + 5
	total	6.0 + 5	2.4 + 5	1.8 + 6	1.3 + 6	4.0 + 5	1.1 + 6

t_{H_2O} = 10 days

Relative Leach Rate	Pathway	Organ: Whole Body	Lung	Total Marrow	Skeleton	LLI	Thyroid
10^5	sh	6.8 + 5	6.8 + 5	6.8 + 5	6.8 + 5	6.8 + 5	6.8 + 5
	dw	1.6 + 7	5.2 + 5	6.0 + 7	1.2 + 8	3.2 + 8	1.4 + 7
	af	2.4 + 6	2.8 + 5	9.2 + 6	2.0 + 7	1.3 + 7	1.5 + 6
	total	1.9 + 7	1.5 + 6	7.0 + 7	1.4 + 8	3.3 + 8	1.6 + 7
10^3	sh	6.0 + 5	6.0 + 5	6.0 + 5	6.0 + 5	6.0 + 5	6.0 + 5
	dw	1.6 + 7	4.5 + 5	5.9 + 7	1.2 + 8	3.0 + 8	1.4 + 7
	af	2.3 + 6	2.8 + 5	9.2 + 6	2.0 + 7	1.2 + 7	1.5 + 6
	total	1.8 + 7	1.3 + 6	6.8 + 7	1.4 + 8	3.1 + 8	1.6 + 7
10^1	sh	9.8 + 4	9.8 + 4	9.8 + 4	9.8 + 4	9.8 + 4	9.8 + 4
	dw	8.9 + 6	8.8 + 4	3.9 + 7	1.0 + 8	4.6 + 7	3.3 + 6
	af	2.0 + 6	2.6 + 5	7.9 + 6	1.7 + 7	2.8 + 6	1.1 + 6
	total	1.1 + 7	4.5 + 5	4.7 + 7	1.2 + 8	4.8 + 7	5.4 + 6
10^0	sh	2.8 + 4	2.8 + 4	2.8 + 4	2.8 + 4	2.8 + 4	2.8 + 4
	dw	3.4 + 6	4.9 + 4	1.5 + 7	4.1 + 7	6.6 + 6	2.0 + 6
	af	1.1 + 6	2.4 + 5	4.1 + 6	7.4 + 6	8.8 + 5	8.4 + 5
	total	4.5 + 6	3.1 + 5	1.9 + 7	4.8 + 7	7.5 + 6	2.8 + 6
10^{-1}	sh	6.6 + 3	6.6 + 3	6.6 + 3	6.6 + 3	6.6 + 3	6.6 + 3
	dw	6.9 + 4	1.5 + 4	2.6 + 5	4.6 + 5	8.4 + 4	6.7 + 4
	af	2.0 + 5	7.7 + 4	5.8 + 5	3.3 + 5	1.1 + 5	2.6 + 5
	total	2.7 + 5	9.8 + 4	8.4 + 5	7.9 + 5	2.0 + 5	3.4 + 5

t_{H_2O} = 10^2 days

Relative Leach Rate	Pathway	Organ: Whole Body	Lung	Total Marrow	Skeleton	LLI	Thyroid
10^5	sh	3.7 + 5	3.7 + 5	3.7 + 5	3.7 + 5	3.7 + 5	3.7 + 5
	dw	1.0 + 7	2.4 + 5	3.7 + 7	6.9 + 7	2.4 + 8	7.3 + 6
	af	1.0 + 6	8.8 + 3	4.2 + 6	1.0 + 7	9.3 + 6	3.1 + 5
	total	1.2 + 7	6.1 + 5	4.2 + 7	8.0 + 7	2.5 + 8	8.0 + 6
10^3	sh	3.4 + 5	3.4 + 5	3.4 + 5	3.4 + 5	3.4 + 5	3.4 + 5
	dw	1.0 + 7	2.2 + 5	3.7 + 7	6.9 + 7	2.3 + 8	7.1 + 6
	af	9.9 + 5	8.1 + 3	4.2 + 6	1.0 + 7	8.9 + 6	3.0 + 5
	total	1.1 + 7	5.7 + 5	4.1 + 7	8.0 + 7	2.4 + 8	7.7 + 6
10^1	sh	4.7 + 4	4.7 + 4	4.7 + 4	4.7 + 4	4.7 + 4	4.7 + 4
	dw	5.1 + 6	2.8 + 4	2.2 + 7	5.6 + 7	3.6 + 7	1.3 + 6
	af	7.3 + 5	1.0 + 3	3.3 + 6	9.0 + 6	1.8 + 6	8.2 + 4
	total	5.9 + 6	7.5 + 4	2.5 + 6	6.5 + 7	3.8 + 7	1.4 + 6
10^0	sh	4.7 + 3	4.7 + 3	4.7 + 3	4.7 + 3	4.7 + 3	4.7 + 3
	dw	1.4 + 6	2.8 + 3	6.4 + 6	1.7 + 7	4.3 + 6	1.9 + 5
	af	2.2 + 5	1.0 + 2	1.0 + 6	2.8 + 6	3.0 + 5	1.8 + 4
	total	1.6 + 6	7.5 + 3	7.4 + 6	2.0 + 7	4.6 + 6	2.1 + 5
10^{-1}	sh	4.7 + 1	4.7 + 1	4.7 + 1	4.7 + 1	4.7 + 1	4.7 + 1
	dw	1.4 + 4	2.8 + 1	6.4 + 4	1.7 + 5	4.3 + 4	1.9 + 3
	af	2.2 + 3	1.0 + 0	1.0 + 4	2.8 + 4	3.0 + 3	1.8 + 2
	total	1.6 + 4	7.5 + 1	7.4 + 4	2.0 + 5	4.6 + 4	2.1 + 3

t_{H_2O} = 10^3 days

Relative Leach Rate	Pathway	Organ: Whole Body	Lung	Total Marrow	Skeleton	LLI	Thyroid
10^5	sh	5.4 + 4	5.4 + 4	5.4 + 4	5.4 + 4	5.4 + 4	5.4 + 4
	dw	9.4 + 5	3.5 + 4	2.6 + 6	1.3 + 6	4.1 + 7	1.3 + 6
	af	3.5 + 4	1.3 + 3	1.0 + 5	4.8 + 4	1.5 + 6	4.8 + 4
	total	1.0 + 6	8.9 + 4	2.8 + 6	1.4 + 6	4.3 + 7	1.4 + 6
10^3	sh	5.3 + 4	5.3 + 4	5.3 + 4	5.3 + 4	5.3 + 4	5.3 + 4
	dw	9.1 + 5	3.3 + 4	2.5 + 6	1.3 + 6	4.0 + 7	1.2 + 6
	af	3.4 + 4	1.2 + 3	9.3 + 4	4.6 + 4	1.5 + 6	4.6 + 4
	total	1.0 + 6	8.7 + 4	2.7 + 6	1.4 + 6	4.1 + 7	1.3 + 6
10^1	sh	7.9 + 3	7.9 + 3	7.9 + 3	7.9 + 3	7.9 + 3	7.9 + 3
	dw	1.4 + 5	5.0 + 3	3.8 + 5	1.9 + 5	6.0 + 6	1.9 + 5
	af	5.0 + 3	1.9 + 2	1.4 + 4	6.9 + 3	2.2 + 5	6.9 + 3
	total	1.5 + 5	1.3 + 4	4.0 + 5	2.0 + 5	6.2 + 6	2.0 + 5
10^0	sh	7.7 + 2	7.7 + 2	7.7 + 2	7.7 + 2	7.7 + 2	7.7 + 2
	dw	1.3 + 4	4.9 + 2	3.7 + 4	1.8 + 4	5.8 + 5	1.8 + 4
	af	4.9 + 2	1.8 + 1	1.4 + 3	6.7 + 2	2.2 + 4	6.7 + 2
	total	1.5 + 4	1.3 + 3	4.0 + 4	2.0 + 4	6.2 + 5	2.0 + 4
10^{-1}	sh	7.7 + 0	7.7 + 0	7.7 + 0	7.7 + 0	7.7 + 0	7.7 + 0
	dw	1.3 + 2	4.9 + 0	3.7 + 2	1.8 + 2	5.8 + 3	1.8 + 2
	af	4.9 + 0	1.8 + 1	1.4 + 1	6.7 + 0	2.2 + 2	6.7 + 0
	total	1.5 + 2	1.3 + 1	4.0 + 2	2.0 + 2	6.2 + 3	2.0 + 2

Table GU.

Resultant Population Doses for a Free-Flowing River System for Releases from the Melt Debris after a PWR-7 Accident

t_{H_2O} = 1 day

Relative Leach Rate	Pathway	Organ: Whole Body	Lung	Total Marrow	Skeleton	LLI	Thyroid
10^5	sh	4.7 + 5	4.7 + 5	4.7 + 5	4.7 + 5	4.7 + 5	4.7 + 5
	dw	8.8 + 6	4.7 + 5	3.3 + 7	7.3 + 7	1.4 + 8	1.4 + 7
	af	3.0 + 6	1.2 + 6	9.3 + 6	1.2 + 7	9.1 + 6	4.1 + 6
	total	1.2 + 7	2.1 + 6	4.2 + 7	8.5 + 7	1.4 + 8	1.9 + 7
10^3	sh	4.2 + 5	4.2 + 5	4.2 + 5	4.2 + 5	4.2 + 5	4.2 + 5
	dw	8.5 + 6	4.2 + 5	3.3 + 7	7.2 + 7	1.2 + 8	1.4 + 7
	af	3.0 + 6	1.2 + 6	9.2 + 6	1.2 + 7	8.0 + 6	4.1 + 6
	total	1.2 + 7	2.0 + 6	4.2 + 7	8.4 + 7	1.3 + 8	1.8 + 7
10^1	sh	9.5 + 4	9.5 + 4	9.5 + 4	9.5 + 4	9.5 + 4	9.5 + 4
	dw	4.9 + 6	2.2 + 5	2.0 + 7	5.3 + 7	1.6 + 7	3.9 + 6
	af	2.5 + 6	1.0 + 6	7.9 + 6	9.6 + 6	2.2 + 6	3.5 + 6
	total	7.6 + 6	1.4 + 6	2.9 + 7	6.3 + 7	1.8 + 7	7.4 + 6
10^0	sh	3.3 + 4	3.3 + 4	3.3 + 4	3.3 + 4	3.3 + 4	3.3 + 4
	dw	1.4 + 6	1.2 + 5	6.1 + 6	1.5 + 7	2.4 + 6	8.0 + 5
	af	1.5 + 6	6.0 + 5	4.5 + 6	4.1 + 6	9.7 + 5	2.5 + 6
	total	2.9 + 6	7.5 + 5	1.1 + 7	1.9 + 7	3.4 + 6	3.3 + 6
10^{-1}	sh	1.4 + 4	1.4 + 4	1.4 + 4	1.4 + 4	1.4 + 4	1.4 + 4
	dw	9.7 + 4	3.6 + 4	3.0 + 5	2.7 + 5	6.9 + 4	1.9 + 5
	af	4.4 + 5	1.9 + 5	1.3 + 6	6.3 + 5	2.4 + 5	8.0 + 5
	total	5.5 + 5	2.4 + 5	1.6 + 6	9.0 + 5	3.3 + 5	1.0 + 6

t_{H_2O} = 10 days

Relative Leach Rate	Pathway	Organ: Whole Body	Lung	Total Marrow	Skeleton	LLI	Thyroid
10^5	sh	3.2 + 5	3.2 + 5	3.2 + 5	3.2 + 5	3.2 + 5	3.2 + 5
	dw	5.6 + 6	2.7 + 5	2.0 + 7	4.1 + 7	1.1 + 8	1.0 + 7
	af	1.2 + 6	2.6 + 5	4.3 + 6	7.1 + 6	4.8 + 6	1.3 + 6
	total	7.1 + 6	8.5 + 5	2.4 + 7	4.8 + 7	1.1 + 8	1.1 + 7
10^3	sh	2.8 + 5	2.8 + 5	2.8 + 5	2.8 + 5	2.8 + 5	2.8 + 5
	dw	5.3 + 6	2.3 + 5	2.0 + 7	4.0 + 7	1.0 + 8	9.9 + 6
	af	1.2 + 6	2.6 + 5	4.3 + 6	7.1 + 6	4.4 + 6	1.3 + 6
	total	6.8 + 6	7.7 + 5	2.4 + 7	4.7 + 7	1.0 + 8	1.1 + 7
10^1	sh	5.1 + 4	5.1 + 4	5.1 + 4	5.1 + 4	5.1 + 4	5.1 + 4
	dw	3.0 + 6	6.2 + 4	1.3 + 7	3.4 + 7	1.5 + 7	2.8 + 6
	af	1.1 + 6	2.5 + 5	3.8 + 6	6.3 + 6	1.1 + 6	9.8 + 5
	total	4.1 + 6	3.6 + 5	1.7 + 7	4.0 + 7	1.6 + 7	3.8 + 6
10^0	sh	2.2 + 4	2.2 + 4	2.2 + 4	2.2 + 4	2.2 + 4	2.2 + 4
	dw	1.2 + 6	4.5 + 4	5.3 + 6	1.4 + 7	2.2 + 6	4.7 + 5
	af	7.3 + 5	2.3 + 5	2.4 + 6	3.0 + 6	4.8 + 5	7.9 + 5
	total	1.9 + 6	2.8 + 5	7.7 + 6	1.7 + 7	2.7 + 6	1.5 + 6
10^{-1}	sh	6.3 + 3	6.3 + 3	6.3 + 3	6.3 + 3	6.3 + 3	6.3 + 3
	dw	4.6 + 4	1.4 + 4	1.5 + 5	1.8 + 5	3.9 + 4	7.0 + 4
	af	1.8 + 5	7.3 + 4	5.4 + 5	2.7 + 5	9.7 + 4	2.5 + 5
	total	2.3 + 5	9.3 + 4	6.9 + 5	4.5 + 5	2.0 + 5	3.2 + 5

t_{H_2O} = 10^2 days

Relative Leach Rate	Pathway	Organ: Whole Body	Lung	Total Marrow	Skeleton	LLI	Thyroid
10^5	sh	1.3 + 5	1.3 + 5	1.3 + 5	1.3 + 5	1.3 + 5	1.3 + 5
	dw	3.3 + 6	8.9 + 4	1.2 + 7	2.3 + 7	7.7 + 7	2.3 + 6
	af	3.3 + 5	3.3 + 3	1.4 + 6	3.5 + 6	3.0 + 6	9.9 + 4
	total	3.8 + 6	2.2 + 5	1.4 + 7	2.6 + 7	8.0 + 7	2.6 + 6
10^3	sh	1.2 + 5	1.2 + 5	1.2 + 5	1.2 + 5	1.2 + 5	1.2 + 5
	dw	3.3 + 6	7.9 + 4	1.2 + 7	2.3 + 7	7.3 + 7	2.2 + 6
	af	3.3 + 5	2.9 + 3	1.4 + 6	3.5 + 6	2.9 + 6	9.6 + 4
	total	3.7 + 6	2.0 + 5	1.3 + 7	2.6 + 7	7.6 + 7	2.5 + 6
10^1	sh	1.5 + 4	1.5 + 4	1.5 + 4	1.5 + 4	1.5 + 4	1.5 + 4
	dw	1.7 + 6	8.9 + 3	7.3 + 6	1.9 + 7	1.1 + 7	4.1 + 5
	af	2.4 + 5	3.3 + 2	1.1 + 6	3.0 + 6	5.7 + 5	2.7 + 4
	total	1.9 + 6	2.4 + 4	8.4 + 6	2.2 + 7	1.2 + 7	4.5 + 5
10^0	sh	1.5 + 3	1.5 + 3	1.5 + 3	1.5 + 3	1.5 + 3	1.5 + 3
	dw	4.7 + 5	8.9 + 2	2.1 + 6	5.7 + 6	1.4 + 6	6.0 + 4
	af	7.4 + 4	3.3 + 1	3.4 + 5	9.3 + 5	9.8 + 4	5.8 + 3
	total	5.5 + 5	2.4 + 3	2.5 + 6	6.6 + 6	1.5 + 6	6.7 + 4
10^{-1}	sh	1.5 + 1	1.5 + 1	1.5 + 1	1.5 + 1	1.5 + 1	1.5 + 1
	dw	4.7 + 3	8.9 + 0	2.1 + 4	5.7 + 4	1.4 + 4	6.0 + 2
	af	7.4 + 2	3.3 − 1	3.4 + 3	9.3 + 3	9.8 + 2	5.8 + 1
	total	5.5 + 3	2.4 + 1	2.5 + 4	6.6 + 4	1.5 + 4	6.7 + 2

t_{H_2O} = 10^3 days

Relative Leach Rate	Pathway	Organ: Whole Body	Lung	Total Marrow	Skeleton	LLI	Thyroid
10^5	sh	1.7 + 4	1.7 + 4	1.7 + 4	1.7 + 4	1.7 + 4	1.7 + 4
	dw	3.0 + 5	1.1 + 4	8.1 + 5	4.0 + 5	1.3 + 7	4.0 + 5
	af	1.1 + 4	4.0 + 2	3.0 + 4	1.5 + 4	4.8 + 5	1.5 + 4
	total	3.2 + 5	2.8 + 4	8.6 + 5	4.3 + 5	1.3 + 7	4.3 + 5
10^3	sh	1.6 + 4	1.6 + 4	1.6 + 4	1.6 + 4	1.6 + 4	1.6 + 4
	dw	2.9 + 5	1.0 + 4	7.9 + 5	3.9 + 5	1.3 + 7	3.9 + 5
	af	1.1 + 4	3.9 + 2	2.9 + 4	1.4 + 4	4.6 + 5	1.4 + 4
	total	3.1 + 5	2.7 + 4	8.3 + 5	4.2 + 5	1.3 + 7	4.2 + 5
10^1	sh	2.5 + 3	2.5 + 3	2.5 + 3	2.5 + 3	2.5 + 3	2.5 + 3
	dw	4.3 + 5	1.6 + 3	1.2 + 5	5.8 + 4	1.9 + 6	5.8 + 4
	af	1.6 + 3	5.8 + 1	4.4 + 3	2.2 + 3	6.9 + 4	2.1 + 3
	total	4.7 + 4	4.0 + 3	1.2 + 5	6.3 + 4	1.9 + 6	6.2 + 4
10^0	sh	2.4 + 2	2.4 + 2	2.4 + 2	2.4 + 2	2.4 + 2	2.4 + 2
	dw	4.2 + 3	1.5 + 2	1.1 + 4	5.7 + 3	1.8 + 5	5.6 + 3
	af	1.5 + 2	5.6 + 0	4.3 + 2	2.1 + 2	6.7 + 3	2.1 + 2
	total	4.6 + 3	3.9 + 2	1.2 + 4	6.3 + 3	1.9 + 5	6.1 + 3
10^{-1}	sh	2.4 + 0	2.4 + 0	2.4 + 0	2.4 + 0	2.4 + 0	2.4 + 0
	dw	4.1 + 1	1.5 + 0	1.1 + 2	5.7 + 1	1.8 + 3	5.6 + 1
	af	1.5 + 0	5.6 − 2	4.2 + 0	2.1 + 0	6.7 + 1	2.1 + 0
	total	4.6 + 1	3.9 + 0	1.2 + 2	6.3 + 1	1.9 + 3	6.1 + 1

Table G9. Health Effects for Releases from the Melt Debris After a PWR-7 Accident[a]

		Cancer Fatalities								Thryoid Effects		Genetic Effects			
Waterbody	t_{H_2O} (days)	Total	Total (sum)	Leukemia	Lung	Breast	Bone	GI	Other	Benign Nodules	Cancers	Dom.	Mul.	Chrom.	Abort.
Lake															
Nearshore	1	74	196	54	4	16	28	79	15	134	191	28	52	3	22
	10	54	155	37	3	11	17	76	11	99	146	20	38	2	16
	10^2	23	88	19	0	5	9	50	5	17	26	9	16	1	17
	10^3	2	9	1	0	0	0	8	0	3	4	1	1	0	1
Offshore	1	620	1394	568	11	131	366	190	128	282	415	234	436	26	183
	10	511	1236	511	4	108	331	177	105	115	170	192	360	21	151
	10^2	268	706	278	1	57	179	136	55	52	77	101	188	11	79
	10^3	5	22	3	0	1	0	18	1	6	9	2	3	0	1
Estuary															
Small	1	2920	8009	1363	289	617	380	4760	600	4425	6415	1099	2050	122	862
	10	2067	6475	880	222	437	159	4352	425	2682	3948	779	1452	87	610
	10^2	1338	4653	596	131	283	104	3264	275	1743	2566	504	939	56	395
	10^3	195	1096	401	15	41	14	585	40	255	375	73	137	8	57
Large	1	1946	6522	1221	111	411	435	3944	400	2950	4343	733	1366	82	574
	10	1094	4783	625	62	231	104	3536	225	1475	2171	412	769	46	323
	10^2	705	3500	398	27	61	149	2720	145	978	1441	266	495	30	208
	10^3	118	636	71	4	25	9	503	24	174	257	44	83	5	35
Ocean															
Nearshore	1	267	1269	182	8	54	69	911	53	415	612	96	1179	11	75
	10	195	991	114	6	41	15	775	40	228	336	73	1137	8	57
	10^2	77	455	48	1	16	7	367	16	107	158	29	54	3	23
	10^3	13	78	8	0	3	1	63	3	19	28	5	9	1	4
Offshore	1	35	110	28	1	7	14	53	7	39	57	13	25	2	10
	10	15	66	9	0	3	2	49	3	19	28	6	10	1	4
	10^2	9	49	6	0	2	1	38	2	12	18	3	6	0	3
	10^3	2	8	1	0	0	0	7	0	2	3	1	1	0	0
River															
Dammed	1	2797	9800	2300	62	591	1104	5168	575	3889	5725	1053	1964	117	826
	10	2310	8438	1988	33	488	966	4488	475	2146	3158	870	1623	97	682
	10^2	1459	5167	1193	14	308	552	3400	300	1073	1579	550	1025	61	431
	10^3	122	728	80	2	26	10	585	25	188	276	46	85	5	36
Free-flowing	1	1459	4463	1193	47	308	587	2028	300	2548	3751	550	1025	61	431
	10	863	2889	682	19	183	331	1496	178	1475	2171	325	606	36	255
	10^2	462	1863	398	5	98	179	1088	95	349	513	174	325	19	136
	10^3	39	221	24	1	8	3	177	8	58	85	15	27	2	12

[a] Instantaneous leaching is assumed.

Table G10. Resultant Whole-Body Population Doses for Releases Into a Lake after Various Categories of Accidents

t_{H_2O}	Pathway	Nearshore Region								Offshore Region							
		PWR-7		PWR-5		PWR-3		PWR-1		PWR-7		PWR-5		PWR-3		PWR-1	
		Debris	Total	Debris	Total	Debris	Total	Debris	Total	Debris	Total	Debris	Total	Debris	Total	Debris	Total
1	sh	1.3 + 5	2.2 + 5	1.3 + 5	2.2 + 5	1.3 + 5	2.0 + 5	8.4 + 4	1.4 + 5	3.3 + 5	2.2 + 6	3.3 + 5	[illegible].1 + 6	3.2 + 5	1.8 + 6	2.7 + 5	1.4 + 6
	dw	4.1 + 5	7.4 + 5	4.1 + 5	7.4 + 5	4.1 + 5	6.7 + 5	3.7 + 5	5.6 + 5	4.4 + 6	7.0 + 6	4.4 + 6	7.0 + 6	4.4 + 6	6.3 + 6	4.2 + 6	5.6 + 6
	af	8.3 + 4	1.2 + 6	8.3 + 4	1.2 + 6	8.3 + 4	9.7 + 5	5.3 + 5	7.2 + 5	4.2 + 5	5.0 + 6	4.2 + 5	5.0 + 6	4.2 + 5	4.1 + 6	4.2 + 5	3.2 + 6
	total	6.7 + 5	2.1 + 6	6.7 + 5	2.1 + 6	6.7 + 5	1.8 + 6	5.8 + 5	1.4 + 5	5.6 + 6	1.4 + 7	5.6 + 6	1.4 + 7	5.6 + 6	1.4 + 7	5.3 + 6	1.0 + 7
10	sh	1.3 + 5	1.8 + 5	1.3 + 5	1.7 + 5	1.3 + 5	1.6 + 5	7.9 + 4	1.1 + 5	1.3 + 5	1.1 + 6	1.3 + 5	1.1 + 6	1.3 + 5	8.7 + 5	1.1 + 5	6.4 + 5
	dw	2.8 + 5	3.8 + 5	2.8 + 5	3.8 + 5	2.8 + 5	3.5 + 5	2.3 + 5	2.8 + 5	3.8 + 6	5.0 + 6	3.8 + 6	5.0 + 6	3.8 + 6	4.7 + 6	3.6 + 6	4.3 + 6
	af	3.3 + 4	3.6 + 5	3.3 + 4	3.5 + 5	3.3 + 4	2.9 + 5	3.1 + 4	2.2 + 5	2.6 + 5	2.3 + 6	2.6 + 5	2.3 + 6	2.6 + 5	1.8 + 6	2.5 + 5	1.4 + 6
	total	5.0 + 5	9.7 + 5	5.0 + 5	9.6 + 5	4.9 + 5	8.6 + 5	3.9 + 5	6.6 + 5	4.6 + 6	8.7 + 6	4.6 + 6	8.7 + 6	4.6 + 6	7.8 + 6	4.4 + 6	6.7 + 6
10^2	sh	1.5 + 4	1.5 + 4	1.5 + 4	1.5 + 4	1.5 + 4	1.5 + 4	9.3 + 3	9.3 + 3	4.0 + 4	4.0 + 4	4.0 + 4	4.0 + 4	3.9 + 4	3.9 + 4	2.4 + 4	2.4 + 4
	dw	1.7 + 5	1.7 + 5	1.7 + 5	1.7 + 5	1.7 + 5	1.7 + 5	1.4 + 5	1.4 + 5	2.1 + 6	2.1 + 6	2.1 + 6	2.1 + 6	2.1 + 6	2.1 + 6	2.0 + 6	2.0 + 6
	af	9.2 + 3	9.3 + 3	9.2 + 3	9.3 + 3	9.1 + 3	9.1 + 3	8.2 + 3	8.2 + 3	8.4 + 4	8.6 + 4	8.4 + 4	8.6 + 4	8.4 + 4	8.4 + 4	8.1 + 4	8.1 + 4
	total	2.2 + 5	2.3 + 5	2.2 + 5	2.3 + 5	2.2 + 5	2.2 + 5	1.8 + 5	1.8 + 5	2.5 + 6	2.5 + 6	2.5 + 6	2.5 + 6	2.5 + 6	2.5 + 6	2.3 + 6	2.3 + 6
10^3	sh	1.7 + 3	1.7 + 3	1.7 + 3	1.7 + 3	1.7 + 3	1.7 + 3	1.0 + 3	1.0 + 3	7.2 + 3	7.4 + 3	7.2 + 3	7.4 + 3	7.2 + 3	7.2 + 3	4.4 + 3	4.4 + 3
	dw	1.4 + 4	1.4 + 4	1.4 + 4	1.4 + 4	1.4 + 4	1.4 + 4	8.4 + 3	8.4 + 3	3.0 + 4	3.0 + 4	3.0 + 4	3.0 + 4	2.9 + 4	2.9 + 4	1.8 + 4	1.8 + 4
	af	3.1 + 2	3.1 + 2	3.1 + 2	3.1 + 2	3.0 + 2	3.0 + 2	1.9 + 2	1.9 + 2	3.3 + 2	3.4 + 2	3.3 + 2	3.4 + 2	3.3 + 2	3.3 + 2	2.0 + 2	2.0 + 2
	total	1.7 + 4	1.7 + 4	1.7 + 4	1.7 + 4	1.6 + 4	1.4 + 4	1.0 + 4	1.0 + 4	3.8 + 4	3.9 + 4	3.8 + 4	3.9 + 4	3.7 + 4	3.7 + 4	2.3 + 4	2.3 + 4

Table G11. Resultant Whole-Body Population Doses for Releases into an Estuary After Various Categories of Accidents

t_{H_2O}	Pathway	Small Estuary								Large Estuary							
		PWR-7		PWR-5		PWR-3		PWR-1		PWR-7		PWR-5		PWR-3		PWR-1	
		Debris	Total	Debris	Total	Debris	Total	Debris	Total	Debris	Total	Debris	Total	Debris	Total	Debris	Total
1	sh	1.1 + 7	4.9 + 7	1.1 + 7	4.9 + 7	1.1 + 7	4.1 + 7	7.8 + 6	3.0 + 7	2.6 + 6	1.4 + 7	2.6 + 6	1.4 + 7	2.6 + 6	1.1 + 7	2.0 + 6	8.0 + 6
	af	1.3 + 7	1.9 + 7	1.3 + 7	1.9 + 7	1.2 + 7	1.7 + 7	8.0 + 6	1.1 + 7	1.3 + 7	1.9 + 7	1.3 + 7	1.9 + 7	1.3 + 7	1.8 + 7	1.0 + 7	1.4 + 7
	total	2.4 + 7	6.8 + 7	2.4 + 7	6.8 + 7	2.3 + 7	5.8 + 7	1.6 + 7	4.1 + 7	1.6 + 7	3.3 + 7	1.6 + 7	3.3 + 7	1.6 + 7	2.9 + 7	1.2 + 7	2.2 + 7
10	sh	9.1 + 6	2.7 + 7	9.1 + 6	2.7 + 7	8.9 + 6	2.3 + 7	6.0 + 6	1.6 + 7	1.9 + 6	7.0 + 6	1.9 + 6	7.0 + 6	1.8 + 6	5.9 + 6	1.3 + 6	4.3 + 6
	af	8.0 + 6	1.0 + 7	8.0 + 6	1.0 + 7	8.0 + 6	1.0 + 7	5.0 + 6	5.0 + 7	7.1 + 6	9.0 + 6	7.1 + 6	9.0 + 6	7.1 + 6	8.1 + 6	4.5 + 6	5.7 + 6
	total	1.7 + 7	3.7 + 7	1.7 + 7	3.7 + 7	1.7 + 7	3.3 + 7	1.1 + 7	2.3 + 7	9.0 + 6	1.6 + 7	9.0 + 6	1.6 + 7	8.9 + 6	1.4 + 7	5.8 + 6	1.0 + 7
10^2	sh	5.7 + 6	5.8 + 6	5.7 + 6	5.8 + 6	5.6 + 6	5.6 + 6	3.5 + 6	3.5 + 6	1.0 + 6	1.1 + 6	1.0 + 6	1.1 + 6	1.0 + 6	1.0 + 6	6.3 + 5	6.3 + 5
	af	5.0 + 6	5.0 + 6	5.0 + 6	5.0 + 6	5.0 + 6	5.0 + 6	3.4 + 6	3.4 + 6	4.8 + 6	4.8 + 6	4.8 + 6	4.8 + 6	4.7 + 6	4.7 + 6	3.0 + 6	3.0 + 6
	total	1.1 + 7	1.1 + 7	1.1 + 7	1.1 + 7	1.1 + 7	1.1 + 7	6.9 + 6	6.9 + 6	5.8 + 6	5.9 + 6	5.8 + 6	5.9 + 6	5.7 + 6	5.7 + 6	3.6 + 6	3.6 + 6
10^3	sh	6.3 + 5	6.5 + 5	6.3 + 5	6.5 + 5	6.3 + 5	6.3 + 5	3.9 + 5	3.9 + 5	1.4 + 5	1.4 + 5	1.4 + 5	1.4 + 5	1.4 + 5	1.4 + 5	8.4 + 4	8.4 + 4
	af	1.0 + 6	1.0 + 6	1.0 + 6	1.0 + 6	1.0 + 6	1.0 + 6	5.9 + 5	5.9 + 5	8.4 + 5	8.6 + 5	8.4 + 5	8.6 + 5	8.3 + 5	8.3 + 5	5.2 + 5	5.2 + 5
	total	1.6 + 6	1.6 + 6	1.6 + 6	1.6 + 6	1.6 + 6	1.6 + 6	9.8 + 5	9.8 + 5	9.8 + 5	1.0 + 6	9.8 + 5	1.0 + 6	9.7 + 5	9.7 + 5	6.0 + 5	6.0 + 5

Table G12. Resultant Whole-Body Population Doses for Releases into an Ocean After Various Categories of Accidents

		Nearshore Region								Offshore Region							
		PWR-7		PWR-5		PWR-3		PWR-1		PWR-7		PWR-5		PWR-3		PWR-1	
t_{H_2O}	Pathway	Debris	Total	Debris	Total	Debris	Total	Debris	Total	Debris	Total	Debris	Total	Debris	Total	Debris	Total
1	sh	5.8 + 4	8.1 + 4	5.8 + 4	8.1 + 4	5.7 + 4	7.7 + 4	3.6 + 4	5.0 + 4	3.9 + 3	8.8 + 3	3.9 + 3	8.8 + 3	3.8 + 3	7.6 + 3	2.5 + 3	5.3 + 3
	af	2.2 + 6	3.5 + 6	2.2 + 6	3.5 + 6	2.1 + 6	3.1 + 6	1.6 + 6	2.3 + 6	2.9 + 5	4.9 + 5	2.9 + 5	4.9 + 5	2.9 + 5	4.8 + 3	2.5 + 5	3.8 + 5
	total	2.2 + 6	3.5 + 6	2.2 + 6	3.5 + 6	2.1 + 6	3.1 + 6	1.7 + 6	2.4 + 6	2.9 + 5	5.0 + 5	2.9 + 5	5.0 + 5	2.9 + 5	4.9 + 3	2.5 + 5	3.9 + 5
10	sh	4.8 + 4	5.6 + 4	4.8 + 4	5.6 + 4	4.8 + 4	5.3 + 4	3.0 + 4	3.3 + 4	3.2 + 3	4.8 + 3	3.2 + 3	4.7 + 3	3.2 + 3	4.4 + 3	2.1 + 3	2.9 + 3
	af	1.6 + 6	2.1 + 6	1.6 + 6	2.1 + 6	1.6 + 6	1.9 + 6	9.0 + 5	1.2 + 6	1.2 + 5	2.0 + 5	1.2 + 5	2.0 + 5	1.1 + 5	1.8 + 5	7.7 + 4	1.3 + 5
	total	1.6 + 6	2.1 + 6	1.6 + 6	2.1 + 6	1.6 + 6	1.9 + 6	9.7 + 5	1.3 + 6	1.2 + 5	2.0 + 5	1.2 + 5	2.0 + 5	1.1 + 5	1.8 + 5	7.9 + 4	1.3 + 5
10^2	sh	2.5 + 3	2.6 + 3	2.5 + 3	2.6 + 3	2.5 + 3	2.5 + 3	1.5 + 3	1.5 + 3	1.6 + 3	1.6 + 3	1.6 + 3	1.6 + 3	1.6 + 3	1.6 + 3	9.8 + 2	9.8 + 2
	af	6.3 + 5	6.4 + 5	6.3 + 5	6.4 + 5	6.3 + 5	6.3 + 5	3.9 + 5	3.9 + 5	7.0 + 4	7.1 + 4	7.0 + 4	7.1 + 4	6.9 + 4	6.9 + 4	4.4 + 4	4.4 + 4
	total	6.3 + 5	6.4 + 5	6.3 + 5	6.4 + 5	6.3 + 5	6.3 + 5	3.9 + 5	3.9 + 5	7.1 + 4	7.3 + 4	7.1 + 4	7.3 + 4	7.0 + 4	7.0 + 4	4.5 + 4	4.5 + 4
10^3	sh	7.7 + 2	7.8 + 2	7.7 + 2	7.8 + 2	7.6 + 2	7.6 + 2	4.7 + 2	4.7 + 2	2.4 + 2	2.5 + 2	2.4 + 2	2.5 + 2	2.4 + 2	2.4 + 2	1.5 + 2	1.5 + 2
	af	1.1 + 5	1.1 + 5	1.1 + 5	1.1 + 5	1.1 + [illegible]	1.1 + 5	6.5 + 4	6.5 + 4	1.2 + 4	1.2 + 4	1.2 + 4	1.2 + 4	1.1 + 4	1.1 + 4	7.0 + 3	7.0 + 3
	total	1.1 + 5	1.1 + 5	1.1 + 5	1.1 + 5	1.1 + 5	1.1 + 5	6.5 + 4	6.5 + 4	1.2 + 4	1.2 + 4	1.2 + 4	1.2 + 4	1.1 + 4	1.1 + 4	7.1 + 3	7.1 + 3

Table G13. Resultant Whole-Body Population Doses for Releases into a River System After Various Categories of Accidents

		Dammed River								Free-flowing River							
		PWR-7		PWR-5		PWR-3		PWR-1		PWR-7		PWR-5		PWR-3		PWR-1	
t_{H_2O}	Pathway	Debris	Total	Debris	Total	Debris	Total	Debris	Total	Debris	Total	Debris	Total	Debris	Total	Debris	Total
1	sh	8.6 + 5	1.8 + 6	8.6 + 5	1.8 + 6	8.5 + 5	1.6 + 6	5.6 + 5	1.1 + 6	4.7 + 5	1.4 + 6	4.7 + 5	1.4 + 6	4.7 + 5	1.2 + 6	3.4 + 5	8.6 + 5
	dw	1.8 + 7	2.6 + 7	1.8 + 7	2.6 + 7	1.8 + 7	2.4 + 7	1.5 + 7	1.9 + 7	8.9 + 6	1.7 + 7	8.9 + 6	1.6 + 7	8.9 + 6	1.5 + 7	7.6 + 6	1.2 + 7
	af	4.1 + 6	4.6 + 7	4.1 + 6	4.6 + 7	4.1 + 6	3.7 + 7	4.0 + 6	2.8 + 7	3.0 + 6	4.3 + 7	3.0 + 6	4.3 + 7	3.0 + 6	3.5 + 7	2.9 + 6	2.7 + 7
	total	2.3 + 7	7.5 + 7	2.3 + 7	7.5 + 7	2.3 + 7	6.4 + 7	2.0 + 7	4.9 + 7	1.3 + 7	6.2 + 7	1.3 + 7	6.2 + 7	1.3 + 7	5.2 + 7	1.2 + 7	4.0 + 7
10	sh	6.7 + 5	1.1 + 6	6.7 + 5	1.1 + 6	6.6 + 5	1.0 + 6	4.2 + 5	6.8 + 5	3.3 + 5	7.7 + 5	3.3 + 5	7.6 + 5	3.3 + 5	6.6 + 5	2.2 + 5	4.6 + 5
	dw	1.6 + 7	1.9 + 7	1.6 + 7	1.9 + 7	1.6 + 7	1.8 + 7	1.3 + 7	1.4 + 7	5.5 + 6	8.3 + 6	5.5 + 6	8.3 + 6	5.5 + 6	7.6 + 6	4.5 + 6	5.9 + 6
	af	2.4 + 6	1.5 + 7	2.4 + 6	1.5 + 7	2.4 + 6	1.2 + 7	2.2 + 6	9.4 + 6	1.2 + 6	1.3 + 7	1.2 + 6	1.3 + 7	1.2 + 6	1.1 + 7	1.2 + 6	8.0 + 6
	total	2.0 + 7	3.5 + 7	2.0 + 7	3.5 + 7	1.9 + 7	3.2 + 7	1.6 + 6	2.5 + 7	7.8 + 6	2.3 + 7	7.8 + 6	2.2 + 7	7.8 + 6	1.9 + 7	6.6 + 6	1.5 + 7
10^2	sh	3.7 + 5	3.8 + 5	3.7 + 5	3.8 + 5	3.7 + 5	3.7 + 5	2.3 + 5	2.3 + 5	1.3 + 5	1.4 + 5	1.3 + 5	1.4 + 5	1.3 + 5	1.3 + 5	8.1 + 4	8.1 + 4
	dw	1.0 + 7	1.0 + 7	1.0 + 7	1.0 + 7	1.0 + 7	1.0 + 7	8.0 + 6	8.0 + 6	3.3 + 6	3.4 + 6	3.3 + 6	3.4 + 6	3.3 + 6	3.3 + 6	2.6 + 6	2.6 + 6
	af	9.9 + 5	1.0 + 6	9.9 + 5	1.0 + 6	9.9 + 5	9.9 + 5	8.9 + 5	8.9 + 5	3.3 + 5	3.4 + 5	3.3 + 5	3.4 + 5	3.3 + 5	3.3 + 5	3.0 + 5	3.0 + 5
	total	1.2 + 7	1.2 + 7	1.2 + 7	1.2 + 7	1.2 + 7	1.2 + 7	9.4 + 6	9.4 + 6	4.2 + 6	4.3 + 6	4.2 + 6	4.3 + 6	4.2 + 6	4.2 + 6	3.4 + 6	3.4 + 6
10^3	sh	5.4 + 4	5.6 + 4	5.4 + 4	5.6 + 4	5.4 + 4	5.4 + 4	3.3 + 4	3.3 + 4	1.7 + 4	1.7 + 4	1.7 + 4	1.7 + 4	1.7 + 4	1.7 + 4	1.0 + 4	1.0 + 4
	dw	9.4 + 5	9.6 + 5	9.4 + 5	9.6 + 5	9.3 + 5	9.3 + 5	5.8 + 5	5.8 + 5	3.0 + 5	3.0 + 5	3.0 + 5	3.0 + 5	3.0 + 5	3.0 + 5	1.8 + 5	1.8 + 5
	af	3.5 + 4	3.5 + 4	3.5 + 4	3.6 + 4	3.5 + 4	3.5 + 4	2.1 + 4	2.1 + 4	1.1 + 4	1.1 + 4	1.1 + 4	1.1 + 4	1.1 + 4	1.1 + 4	6.7 + 3	6.7 + 3
	total	1.0 + 6	1.0 + 6	1.0 + 6	1.1 + 6	1.0 + 6	1.0 + 6	6.3 + 5	6.3 + 5	3.2 + 5	3.3 + 5	3.2 + 5	3.3 + 5	3.2 + 5	3.3 + 5	2.0 + 5	2.0 + 5

Table G14. Resultant Population Doses for Releases After a PWR-7 Accident

Whole-Body Population Doses in Person-Rems

Waterbodies	t_{H_2O} (days)	Dose[a]	
		Debris	Total
Free-flowing river + estuary + ocean	1	4 + 7	1 + 8
	10	3 + 7	6 + 7
	10^2	2 + 7	2 + 7
	10^3	2 + 6	2 + 6
Heavily dammed river + estuary + ocean	1	5 + 7	1 + 8
	10	4 + 7	5 + 7
	10^2	2 + 7	2 + 7
	10^3	3 + 6	3 + 6
Large lake + large lake + estuary + ocean	1	1 + 7	2 + 7
	10	1 + 7	1 + 7
	10^2	5 + 6	5 + 6
	10^3	4 + 4	4 + 4

[a] For comparison, see the PWR-7 doses in Tables G10-G13.

Table G15. "Average" Individual Doses Due to Releases from the Melt Debris after a PWR-7 Accident [a,b]

t_{H_2O} = 1 day

Waterbody	Dose (rem) dw	af	sh	Total
Large Lake				
Nearshore	4.3	0.95	0.17	5.4
Rest of Lake	0.40	0.60	0.03	1.0
Estuary				
Small	-	4.6	3.2	7.8
Large	-	1.0	0.47	1.5
Ocean Coast				
Nearshore	-	0.58	0.01	0.59
Offshore	-	0.12	-	0.12
River System				
Dammed	5.3	6.2	0.25	12.0
Free-flowing	2.4	4.2	0.13	6.7

t_{H_2O} = 10 days

Waterbody	Dose (rem) dw	af	sh	Total
Large Lake				
Nearshore	2.9	0.38	0.17	3.4
Rest of Lake	0.35	0.37	0.01	0.73
Estuary				
Small	-	2.9	2.7	5.6
Large	-	0.55	0.34	0.89
Ocean Coast				
Nearshore	-	0.42	0.01	0.43
Offshore	-	0.05	-	0.05
River System				
Dammed	4.7	3.6	0.20	8.5
Free-flowing	1.5	1.7	0.09	3.3

t_{H_2O} = 100 days

Waterbody	Dose (rem) dw	af	sh	Total
Large Lake				
Nearshore	1.8	0.11	0.02	1.9
Rest of Lake	0.19	0.12	0.00	0.31
Estuary				
Small	-	1.8	1.7	3.5
Large	-	0.37	0.18	0.55
Ocean Coast				
Nearshore	-	0.17	0.00	0.17
Offshore	-	0.03	-	0.03
River System				
Dammed	2.9	1.5	0.11	3.5
Free-flowing	0.89	0.46	0.04	1.4

t_{H_2O} = 1000 days

Waterbody	Dose (rem) dw	af	sh	Total
Large Lake				
Nearshore	0.15	0.00	0.00	0.16
Rest of Lake	0.00	0.00	0.00	0.00
Estuary				
Small	-	0.36	0.19	0.55
Large	-	0.06	0.03	0.09
Ocean Coast				
Nearshore	-	0.03	0.00	0.03
Offshore	-	0.00	-	0.00
River System				
Dammed	0.28	0.05	0.02	0.35
Free-flowing	0.08	0.02	0.01	0.11

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[a] See footnotes "a" and "c" of Table 6.5.

[b] For all waterbodies except river systems, the maximum values of the "average" doses are equal to the average values of the "average" doses.

Table G16. "Average" Individual Doses Due to All Possible Direct Releases to the Groundwater after a PWR-7 Accident[(a)]

t_{H_2O} = 1 day

Waterbody	Dose (rem) dw	af	sh	Total
Large Lake				
Nearshore	7.8	14.0	0.28	22.0
Rest of Lake	0.64	7.1	0.20	7.9
Estuary				
Small	-	6.8	14.0	21.0
Large	-	1.5	2.5	4.0
Ocean Coast				
Nearshore	-	0.92	0.02	0.94
Offshore	-	0.20	-	0.20
River System				
Dammed	7.6	70.0	0.53	78.
Free-flowing	4.6	60.0	0.38	65.

t_{H_2O} = 10 days

Waterbody	Dose (rem) dw	af	sh	Total
Large Lake				
Nearshore	4.0	4.1	0.23	4.3
Rest of Lake	0.45	3.3	0.10	3.9
Estuary				
Small	-	3.6	7.9	12.0
Large	-	0.69	1.3	2.0
Ocean Coast				
Nearshore	-	0.55	0.01	0.56
Offshore	-	0.08	-	0.08
River System				
Dammed	5.6	23.0	0.32	29.0
Free-flowing	2.2	18.0	0.21	20.0

t_{H_2O} = 100 days

Waterbody	Dose (rem) dw	af	sh	Total
Large Lake				
Nearshore	1.8	0.11	0.02	1.9
Rest of Lake	0.19	0.12	0.00	0.31
Estuary				
Small	-	1.8	1.7	3.5
Large	-	0.37	0.18	0.55
Ocean Coast				
Nearshore	-	0.17	0.00	0.17
Offshore	-	0.03	-	0.03
River System				
Dammed	2.9	1.5	0.11	3.5
Free-flowing	0.89	0.46	0.04	1.4

t_{H_2O} = 1000 days

Waterbody	Dose (rem) dw	af	sh	Total
Large Lake				
Nearshore	0.15	0.00	0.00	0.16
Rest of Lake	0.00	0.00	0.00	0.00
Estuary				
Small	-	0.36	0.19	0.55
Large	-	0.06	0.03	0.09
Ocean Coast				
Nearshore	-	0.03	0.00	0.03
Offshore	-	0.00	-	0.00
River System				
Dammed	0.28	0.05	0.02	0.35
Free-flowing	0.08	0.02	0.01	0.11

[a] See footnotes in Table G15.

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